Province of British Columbia

Ministry of Environment

KOOCANUSA RESERVOIR
STATE OF THE AQUATIC ENVIRONMENT
1972 - 1988

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Cranbrook

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Mr. Hal Hamilton, Project Management
Dr. Larry Linton, Data Analysis and Interpretation
Dr. Pat Chow-Fraser, Biological Interpretation
Dr. Barry Taylor, Report Editing
Mona Ha, Data Technologist

In addition, Mr. Dave Fernet of Environmental Management Associates was responsible for the fishery analysis.

1.0 BACKGROUND AND INTRODUCTION

1.1 History and Description of Reservoir

From its headwaters in the Rocky Mountains of British Columbia, the Kootenay River (spelled Kootenai in the U.S.A.) runs west to the Rocky Mountain trench and follows it south across the Canada/U.S.A. border into Montana. From there, it flows in a northwesterly direction back across the border into the southern end of Kootenay Lake. In 1972, a dam was constructed on the Kootenai River approximately 27 km upstream from the town of Libby, Montana (Figure 1.1). This impoundment extends from Wardner, British Columbia to the Libby Dam and is known as Lake Koocanusa; it started to fill in March 1972 and reached full pool in June 1974 (Storm et al., 1982).

The reservoir is long and narrow, and when full is roughly divided in half by the International Border. Since the lake's surface area accounts for less than 1% of the drainage basin area, the water budget is controlled almost exclusively by inflow from the three tributaries, the Kootenay River, the Elk River and the Bull River (see Section 3.1). Morphometric and hydrologic characteristics of Lake Koocanusa are available from several sources and we have presented typical values in Table 1.1.

The reservoir undergoes very predictable annual fluctuations in lake elevation (see Section 3.1), a result of management primarily for power generation and flood control. The first turbines were commissioned in 1975 and a selective withdrawal system became operational in 1977. This system permits withdrawal of water from the reservoir at any elevation and thus allows for regulation of the temperature and dissolved oxygen content of outflow.

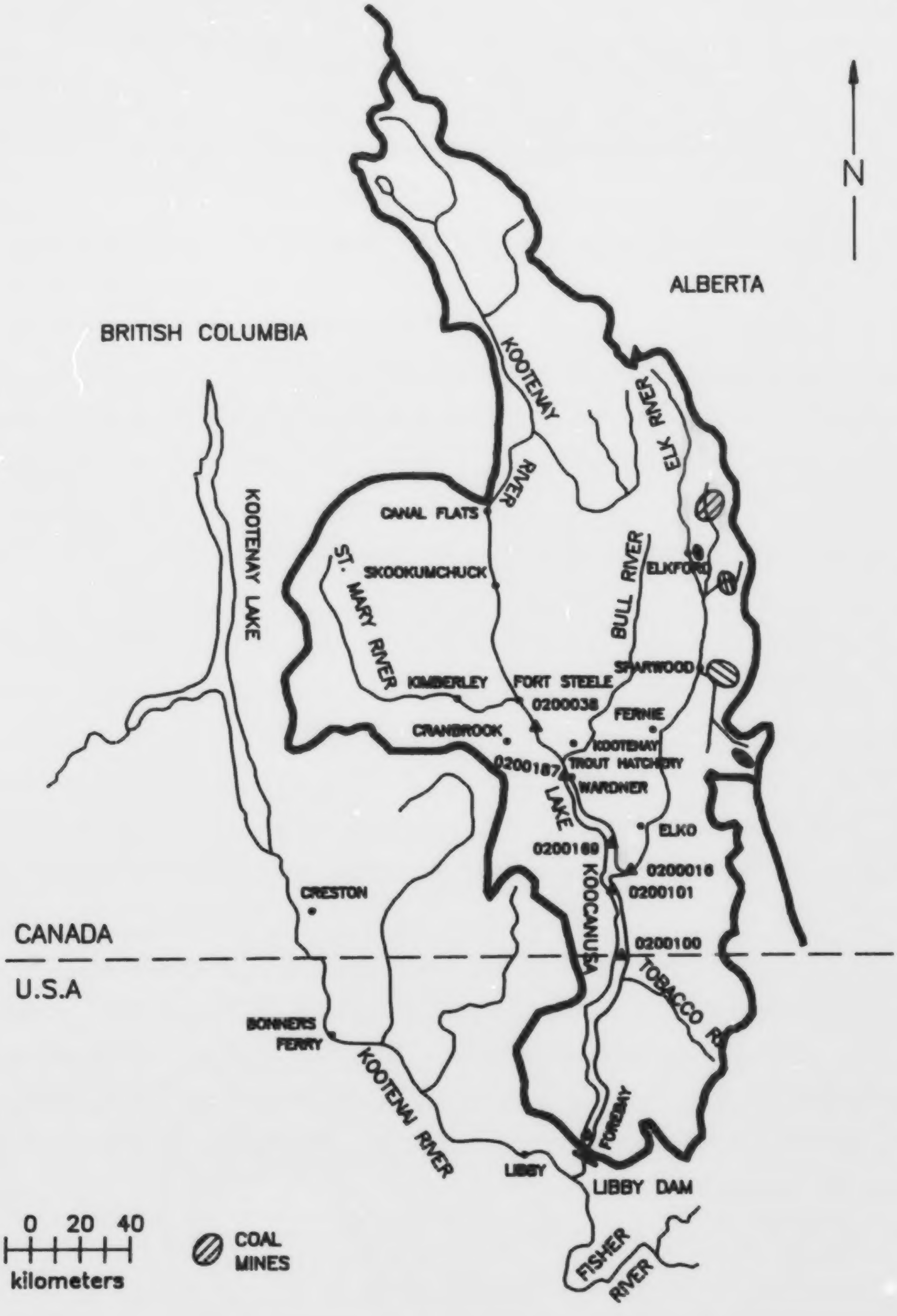
N
ALBERTA
BRITISH COLUMBIA
KOOTENAY RIVER
ELK RIVER
KOOTENAY LAKE
CANAL FLATS
ST. MARY RIVER
SKOOKUMCHUCK
BULL RIVER
ELKFORD
SPARWOOD
KIMBERLEY
FORT STEELE
0200038
FERNIE
CRANBROOK
KOOTENAY TROUT HATCHERY
0200187
WARDNER
ELKO
LAKE KOOCANUSA
0200189
0200016
0200101
CRESTON
CANADA
0200100
U.S.A
TOBACCO R.
BONNERS FERRY
KOOTENAI RIVER
FOREBAY
LIBBY
LIBBY DAM
FISHER RIVER
0 20 40
kilometers
COAL MINES

FIGURE 1.1

KOOTENAY RIVER BASIN
SAMPLE SITES

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Table 1.1 Summary of Morphometric & Hydrologic Characteristics for Lake Koocanusa. Sources include: Bonde and Bush 1975; Chisolm et al, 1989; Crozier and Nordin, unpublished; Daley et al. 1981; Storm et al. 1982.

Variable	
Drainage basin area (km2)	25000
Lake surface area (km2)	
Maximum pool	188
Minimum pool	59
Lake volume (km3)	
Maximum pool	7.2
Minimum pool	1.09
Lake elevation (m)	
Maximum pool	749.5
Minimum pool	697.1
Minimum pool (dead storage)	671.2
Reservoir length (km)	
Maximum pool	155
Minimum pool	68
Maximum width (km)	3
Depth (m)	
Maximum	107
Mean	38
Maximum depth at border (1972–1987)	46
Drawdowm (m)	
Maximum	52
Average (1972–1978)	30
Mean residence time (y)	0.55
Average annual discharge (m3/s)	
Pre-dam (1911–1972)	343
Post-dam (1972–1986)	304.2

1.2 Predicted Water Quality in the Reservoir

Before the dam was built, Bonde and Bush (1975) predicted that the reservoir would become eutrophic because of high nutrient loading from industrial waste discharged to the Kootenay River (based on Vollenweider's (1975) nutrient-loading model). Although visible blooms of the blue-green alga Aphanizomenon flos-aquae occurred in 1974 and 1975, primary production rates in the reservoir from 1972 to 1978 were generally in the range that would characterize it as oligotrophic (Woods, 1979; Crozier and Nordin, unpublished). Therefore, the anticipated development of eutrophy in Lake Koocanusa did not take place. On the contrary, because of substantial reductions in nutrient loading to the Kootenay River from municipal and industrial sources during the late 1970's, there is at present a concern that the nutrient-poor discharge from the reservoir is limiting phytoplankton production further downstream in Kootenay Lake (Daley et al., 1981).

1.3 Watershed Activities

The major urban developments within the Kootenay River drainage basin north of Lake Koocanusa are Cranbrook, Kimberley, Fernie, Sparwood and Elkford (Table 1.3.1; Figure 1.1; Rocchini, 1976 a, b). The economy of the area is heavily based on forestry, mining (metals and coal), recreation, and agriculture (hay and cattle). The amount of industrial and municipal wastes entering the watershed has declined sharply over the eighteen years since the reservoir was created in 1972 (Table 1.3.2). Two previously important industrial sources were: 1) Cominco Sullivan Mine and ammonium phosphate fertilizer plant near Kimberley, which until 1975 discharged large quantities of phosphorus, as well as fluoride, ammonia and some metals into the St. Mary River, a tributary of the Kootenay (Crozier and Nordin, 1981); and 2) Crestbrook Forest Industries Ltd., a bleached kraft pulp mill located near Skookumchuck, which discharged continuously into the Kootenay River until 1981, and was responsible for increased colour downstream in the 1970's (Rocchini, 1981; Figure 1.1). Subsequently,

Table 1.3.1 Major Municipalities and Populations in Watershed.

Municipality	Approximate Population
Cranbrook	18,000
Kimberley	7,000
Fernie	6,000
Sparwood	4,000
Elkford	3,000
Canal Flats	<500
Wardner	<500
Elko	<500

Table 1.3.2 Summary of Changes in the Watershed from 1973 to 1989
STP = Sewage Treatment Plant

Source	Year	Event
Cominco Sullivan Mine, Concentrator, Fertilizer Plant (St. Mary River)	1975	Gypsum pond recycling commenced. (Greatly reduced phosphorus discharge to St. Mary R.)
	1979	Drainage water treatment began. (A.M.D. treatment)
	1987	Fertilizer plant closed. (P. discharge ceased)
Crestbrook Pulpmill (Kootenay River)	1973–1977	Direct discharge of aerated (secondary) treated effluent.
	1977–1980	Pilot rapid infiltration began resulting in periodic reduced discharge.
	1981	R I basins in operation, direct discharge only during spring freshet.
	1986	Basins began to leak ~ 10% of flow to river.
	1989	Direct discharge to river in spring ceased.
City of Cranbrook STP (Joseph Creek,tributary of St. Mary River)	1976	Stopped direct discharge to creek.
	1977	Spray irrigation system for municipal waste water.
City of Kimberley STP (St. Mary River)	1979	New treatment plant began.
Surface coal mining (Elk River)	1975	Fording Coal increased explosive use and switched from anfo to slurry gel formulation
	1980–1989	Two new mines come on line, valley production increases from ~ 9 x 10^6t/y to ~ 17 x 10^6t/y.

significant reductions have been made in nutrient and colour loadings to the river, and these are summarized in Table 1.3.2. Municipal sources include the City of Cranbrook, City of Kimberley, and the municipalities of Elkford, Sparwood and Fernie (Figure 1.1; Table 1.3.1). The largest community, Cranbrook, ceased sewage discharge to the basin in 1976 and now employs a spray irrigation system. The municipalities of Kimberley, Fernie and Sparwood continue to discharge secondarily treated sewage directly to the St. Mary and Elk Rivers.

1.4 Assessment Program

A water quality study was undertaken in 1967 by the U.S. Corps of Army Engineers to assess the effects of dam construction on downstream reaches of the Kootenai River (Bonde and Bush, 1975). The British Columbia Pollution Control Branch (now Waste Management Branch) undertook a similar water quality study of the Kootenay River and its tributaries in Canada (Crozier and Leinweber, 1975; Rocchini, 1976 a, b; Nordin and Crozier, 1976). After the impoundment was completed, these studies were continued by the U.S. Corps of Army Engineers, B.C. Waste Management Branch, with emphasis on the trophic status and water quality of Lake Koocanusa. Eight sampling stations were established on the lake (four in each country) along with several stations on the Kootenay River and its tributaries upstream of the reservoir (Figure 1.1). The station at the border is monitored by both American and Canadian agencies to provide a basis for comparison of results. This station has also been monitored continuously for the longest period.

1.5 Report Objectives and Layout

The Kootenay Region Waste Management Office has requested analysis of the extensive water quality data that have been collected by the American and Canadian monitoring agencies. The objectives of the analysis is:

- to compile all water quality and biological data pertinent to the reservoir and its tributaries (St. Mary and Kootenay Rivers) and incorporate these into an organized and documented database;

- to document the historic water quality sampling and analytic methods;

- to assess compatibility of various information sources, considering changes in detection limits that may have occurred since data collection started in 1972, and to determine their influence upon statistical analysis;

- to identify, describe and interpret spatial and temporal trends and changes in reservoir water quality from natural and anthropogenic factors including:

    - the natural changes that occur as the reservoir matures after filling,

    - impacts on water quality of tributaries and the reservoir from municipal and industrial discharge,

    - anticipated improvements in reservoir water quality after cessation of industrial discharges and introduction of effluent treatment facilities;

- to identify and describe trends and historical changes in phytoplankton and zooplankton community structure, density and diversity;

- to interpret plankton population dynamics with respect to their influence upon fish populations;

- to assess the effects of impoundment and pollution control measures on the water quality and productivity of Kootenay Lake;

- to make recommendations for subsequent water quality monitoring programs that will ensure adequate data for future assessment of water quality.

The following nine sections of this report are each designed to answer one or more of the stated objectives. Section 2 contains a detailed review of the water quality assessment programs that have been carried out on the reservoir. Section 3 examines the physical environment of the reservoir, namely, morphometric, hydrologic and stratification characteristics of Lake Koocanusa. Section 4 summarizes loadings of nutrients, metals and other water quality variables in the two main tributaries, the Elk River and Kootenay River. Section 5 examines the chemical environment in the reservoir with respect to nutrient loading, concentrations of the major ions, metals, nutrients, and other water quality variables. Section 6 deals with the lake biota, including phytoplankton, zooplankton and fish. Section 7 details the trophic status of the reservoir, while Section 8 examines compliance with B.C. provincial water quality criteria. We present our conclusions and recommendations in Sections 9 and 10, respectively. Appendices include a printout and summary statistics from the water quality database.

2.0 WATER QUALITY ASSESSMENT PROGRAMS

2.1 Overview of Canadian Monitoring Program

2.1.1 Field Sampling

Four lake sites and two river sites have been sampled at various times by B.C. Environment (Figure 1.1; Table 2.1.1); for example, the Elk River site (#0200016 or simply #016) has been sampled since 1968, whereas the site on Lake Koocanusa south of Wardner (#187) was only sampled during years of high reservoir elevation between 1974 and 1980.

Prior to April 1988, monthly grab samples from the two tributaries were collected and analyzed by B.C. Environment laboratory (Vancouver); after 1988, biweekly samples were taken from the Kootenay River (site #038) by B.C. Environment, and from the Elk River (site #016) by Environment Canada (lay sampler). Since 1988 all samples have been analyzed by the Canada Centre for Inland Waters in Burlington. Field measurements were taken for temperature, dissolved oxygen and pH using hand held thermometers, a YSI Model S4 DO meter and a Metrohm Herisau Model 8488 pH meter.

During each sampling year, lake sites were sampled approximately monthly from the time of filling (May or June) to the beginning of drawdown (October or November). At these stations, water samples were collected at the surface, middle and bottom of the water column, from 1973 to 1981, and thereafter at the surface and bottom only, using a Van Dorn bottle (7.5 L). Samples were analyzed for nutrients, major ions, heavy metals such as copper (Cu), lead (Pb), zinc (Zn) and manganese (Mn), and other water-quality parameters such as total alkalinity and total hardness. The major nutrients analyzed were total phosphorus (TP), soluble reactive phosphorus (SRP), total ammonia (NH_3), total Kjeldahl nitrogen (TKN), nitrate (NO_3) and nitrite (NO_2). The maximum

Table 2.1.1 Years Sampled at Each Site.

Site #	Site	Latitude	Longitude	Years Sampled
0200016	Elk River at Philips Canyon	49 12 52	115 06 37	1968–1989
0200038	Kootenay River at Picture Valley	49 31 28	115 32 52	1971–1989
0200100	Lake Koocanusa at Border	48 59 55	115 10 48	1972–1989
0200101	Lake Koocanusa off Elk River	49 09 47	115 14 37	1972–1980*
0200169	Lake Koocanusa off Kikomun Cr.	49 15 40	115 16 20	1973–1980*
0200187	Lake Koocanusa south of Wardner	49 24 39	115 25 06	1974–1980*
12301919	Lake Koocanusa at Forebay	48 24 43	115 18 33	1972–1989
12300110	Lake Koocanusa at International Boundary	48 59 44	115 10 43	1972–1989
12301600	Lake Koocanusa below Pincher Creek	48 49 37	115 15 58	1972–1976*
12301830	Lake Koocanusa at Ten Mile Creek	48 35 06	115 13 52	1972–1989*

* Data not analysed in this report.

recorded depth of sampling varied for each station as follows:

#100 (Border)	47 m
#101 (off Elk River)	26 m
#169 (off Kikomun Creek)	21 m
#187 (south of Wardner)	13 m

Field measurements of temperature, dissolved oxygen (DO), pH, specific conductance (SC) and redox potential were taken with a Hydrolab 6 DX-1-160 Surveyor (Hydrolab Corporation, Austin, Texas) until 1983 and with a Hydrolab Surveyor II model 9000 from 1984 to 1989. Prior to 1976, readings were taken at 10 ft intervals in the water column and thereafter at 5 m intervals. Secchi depth transparency and the barometric pressure were also recorded on each sampling occasion.

Aliquots of water were collected from 1, 5, 10, 15 and 20 m (whenever possible) at the Border site for estimation of chlorophyll a concentration (Chla). Monthly samples were also collected at 5-m intervals from surface to 20 m and pooled to form a 1-L composite sample for phytoplankton identification and cell counts (Phyto). Samples were preserved immediately in the field in 5% formalin until 1985 and in 1% (acid) Lugol's solution from then on.

Zooplankton have been collected in vertical hauls with a simple conical net (at a speed of 0.3 m/s) at roughly monthly intervals at the Border site since 1973. The mesh size of these nets has varied from 73 to 158 μm as follows: 153 μm in 1973; 158 μm from 1974-1978; 153 μm from 1979-1984 and 73 μm from 1985 to present. Although difference in mesh size may affect net efficiency, it should not have any effect on size of organisms retained since all of the organisms of interest are greater than 500 μm diameter. The 73 μm mesh has been used to capture rotifers, but since there are only three years' data, they have been excluded from this report. Net diameter was 11.75 cm in 1973-74, and 50 cm thereafter.

Until 1978, vertical tows were taken from various depths between mid-depth and the bottom. After 1978, care was taken to start the haul at a depth of 1 m above the sediment and to end at the surface. This has been done to ensure the capture of Mysis relicta, should it be introduced into the reservoir. Since the majority of planktonic animals are likely to be in the upper few metres of water, the data should be comparable between studies. Samples were preserved in the field in 5% formalin until 1985 and in 50% ethanol/methanol after this date.

2.1.2 Laboratory Techniques

Inductively coupled plasma atomic emission spectrophotometry (ICP) was used initially to scan samples for a wide range of metals at a sensitivity of 0.01 mg/L; samples were subsequently reanalyzed for specific metals with other procedures that had lower detection limits (Appendices 1 and 2). Users of the B.C. Environment database, SEAM, (and its predecessor EQUIS) where all these data are stored, should carefully scrutinize its data output since values from all procedures are intermixed in the database and the code that identifies the respective methods is not used consistently in the older database (EQUIS).

Samples for chlorophyll a were filtered through a 0.45 μm membrane filter and preserved with magnesium carbonate, during vacuum filtration at 15 psi with a 1/6 HP electric pump. Samples were frozen and transported to the laboratory on silica gel in a thermos. Analyses were performed within two months, and the samples were kept frozen in the interval. Samples taken in 1973 and 1974 were analyzed by Western Botanical Services Ltd., Vancouver, B.C. with a fluorometric technique (Strickland and Parsons, 1965) and reported as corrected Chl a values. Samples taken after 1974 were analyzed by B.C. Environment in the Vancouver laboratory with a trichromatic spectrophotometric method (McQuaker, 1973). From 1975 to 1978, phaeophytin values have not been measured (Table 2.1.2) as they were nearly always below detection limits.

Algae were identified to species for the dominant taxa and to genus for the remainder, and enumerated with the aid of (an ICM 405 Carl Zeiss inverted phase-contrast microscope. Samples were sedimented overnight to determine total plankton numbers. Diatoms were digested in hot HNO_3. Since presentation of cell counts alone would bias the analysis of community structure in favour of smaller algal taxa, we have generated crude estimates of algal biomass by applying corresponding biovolume conversions for each species identified in Crozier and Nordin (unpublished) for phytoplankton in Lake Kootenay. In applying a single conversion factor across all years, and all months of the year, we are ignoring seasonal differences in cell size, but we feel that this error is less objectionable than presenting cell counts when one of the study objectives is to evaluate energy flow between plankton compartments in Lake Koocanusa.

Zooplankton were identified to species for the dominant taxa, and to genus for all others, with the aid of a dissecting microscope, and were grouped according to developmental stage (subadults and adults for cladocerans; nauplii and copepodites for copepods). Biomass estimates of zooplankton were calculated using conversion factors presented in Crozier and Duncan (1984). Because of differences among years with respect to length of net hauls (see Field Sampling), expression of animal biomass on a volumetric basis would underestimate populations in the more recent samples; therefore the data are presented on an areal basis. These biomass conversions were done by L.E. McDonald, B.C. Ministry of Environment, Cranbrook.

2.2 Statistical Analyses and Data Presentation

The statistical package used for performing the time-series statistical analyses was WQSTAT (Philips, 1988), a package designed specifically for applications in water-quality statistics. Because most time-series statistical procedures assume that samples being analyzed have been taken at equally spaced intervals, missing samples can seriously impair the accuracy of the test statistic. WQSTAT overcomes these problems by casting the data into equal-length time periods, taking the mean of the samples within each period, and then centering the mean within the period. For example, WQSTAT allows the user to select time periods that are either one month or three months in length. If the three-month period is chosen, then readings for January through March are averaged and the average is placed at February 15. Then the readings for the next three months of the year are averaged and the average is placed at May 15. This procedure is continued for all years spanned by the data, and subsequent analyses are performed on the equally spaced means. If the monthly averaging option is chosen, then means are taken over each month and centered at the 15th of the month.

The choice between procedures must be based upon the frequency of sampling. In the Kootenay and Elk Rivers, samples were taken at roughly monthly intervals, so the monthly averaging option was chosen to increase the sample size. The Lake Koocanusa data do not span the entire year, so the quarterly averaging method was used. For some variables, there were no available data the first quarter of the year, which made it necessary to copy the first reading from the second quarter into the first quarter to allow the program to proceed.

Two forms of time-series analysis were performed using the statistical package WQSTAT II (Phillips, 1988). Nonparametric, seasonal, Kendall's Tau was used to test for long-term linear trends and to estimate the slopes of statistically significant trend lines. Usually the entire data record was used for this analysis, with the following exceptions:

1. If the data from analytic techniques with different detection limits were intermixed, the data for the least sensitive methods were removed.

2. If detection limits changed over the period of record then the test would detect this as a (downward) trend, so only the data from the more recent, usually more sensitive, method were analyzed.

3. In the early years of reservoir filling a number of water quality parameters were erratic and high in Lake Koocanusa, but became more stable and lower in the later record. For such data the trend test would indicate a significant downward trend if all the data were tested; however, the trend in the later data was of greater interest so only these data were analyzed. In all instances, the time period over which the trend test was applied is indicated in the tables of test results.

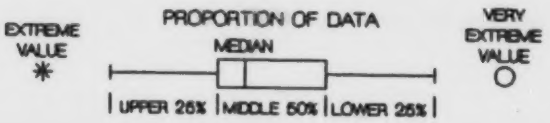
4. If graphical examination of the data indicated nonlinearities in the time series that made application of the test inappropriate, it was not used.

Occasionally, plots of the data showed adjacent time periods between which there were obvious differences in concentration. In these instances the Wilcoxin signed rank test (Phillips, 1988) was used to estimate the magnitude of the difference between medians of the two groups. Used in this way, the technique is not performing a test of a formal statistical hypothesis, but rather is a measuring tool to estimate the magnitude of the difference between medians.

The immense size of water-quality data files provided by the B.C. Ministry of Environment and the U.S.G.S. caused special data-management problems. Firstly, the files were of a totally different format and on different media (5 /14" floppy disks readable by microcomputer versus 2400 foot mag-tapes readable by IBM mainframes, respectively). The U.S.G.S. data were taken to the University of Calgary where personnel from Academic Computing Services were contracted to extract the data, write programs to reformat it to the B.C. Environment format and download it to microcomputer. The resulting files were much too large to be efficiently handled by commercially available data management packages, so it was necessary to write our own applications program to extract the desired data. The program extracts data in two formats, the first format presents each reading for a variable as a separate record along with information regarding analytic codes, depth, detection limits, etc. The second format extracts data for a number of variables measured on the same date and at the same depth and places the readings on a single output record.

Water quality data reported to be less than the detection limit were set to the detection limit for use in statistical procedures. Chlorophyll a concentrations from samples collected at 5 m depth intervals, were averaged over the entire depth before statistical analysis. Figures, correlation analysis, and data recalculation were done using SYGRAPH, GPLOT, and SYSTAT.

We present much of the raw data in the form of Box-and-Whisker plots (see example below). The box itself spans the central 50% of the data, with a line across it at the median. The whiskers indicate the range of the data beyond the ends of the box (i.e. the remaining upper and lower 25% of the data), however they are never larger than 1.5 times the box length. Data beyond the ends of the whiskers indicated as asterisks are beyond 1.5 times the box length from the end of the box and those indicated with an open circle are beyond three times the box length from the box. These are frequently termed "extreme values" and "very extreme values" respectively, and are generally outliers.



3.0 PHYSICAL ENVIRONMENT IN THE RESERVOIR

3.1 Hydrology

Water levels in the reservoir undergo dramatic annual fluctuations, which are controlled by two factors: spring inflow volumes from the three tributaries, Kootenay River (62% of the mean annual inflow), Elk River (26%), and Bull River (11%; Crozier and Nordin, 1981; Figure 3.1.1), and the rate of drawdown in the fall (Figure 3.1.2). Except for the first two years of operation, the main hydrologic characteristic of the lake is a reciprocal pattern of inflow and outflow volumes. In a detailed analysis of the hydrologic budget of Lake Koocanusa from 1983 to 1987, Chisholm et al. (1989) verified that reservoir inflow equalled the annual outflow volume from Libby Dam, except in 1985 when the reservoir did not refill.

The reservoir surface has attained a maximum elevation of 749.5 m in 10 of the 14 years between 1974 and 1987 and typically remains at this elevation for approximately 42 days of the season (Chisholm et al., 1989). In a year of average runoff, the reservoir will be filled by August, and drawdown commences in November and continues until April the following year (Figure 3.1.2). After maximum drawdown, reservoir volume is reduced by up to 85%, mean depth by 51%, surface area by 69% and the total length by 53% (Table 1.1). In the first two years of operation, reservoir elevations were abnormally low and are seen as outliers in Figure 3.1.2. The Canadian portion of the reservoir experiences the largest relative changes in water levels, as illustrated by the area/capacity curve (Figure 3.1.3). Depth at the border site may exceed 40 m at full pool, but when drawdown is maximum (52 m), the reservoir pool begins south of the International Border.

There is some indication that the year-end withdrawal rate has increased since 1983, a trend which may lead to reduced flexibility for controlling temperature of the outfall during drawdown (Chisholm et al., 1983). Depth of withdrawal from 1983 to 1987 has

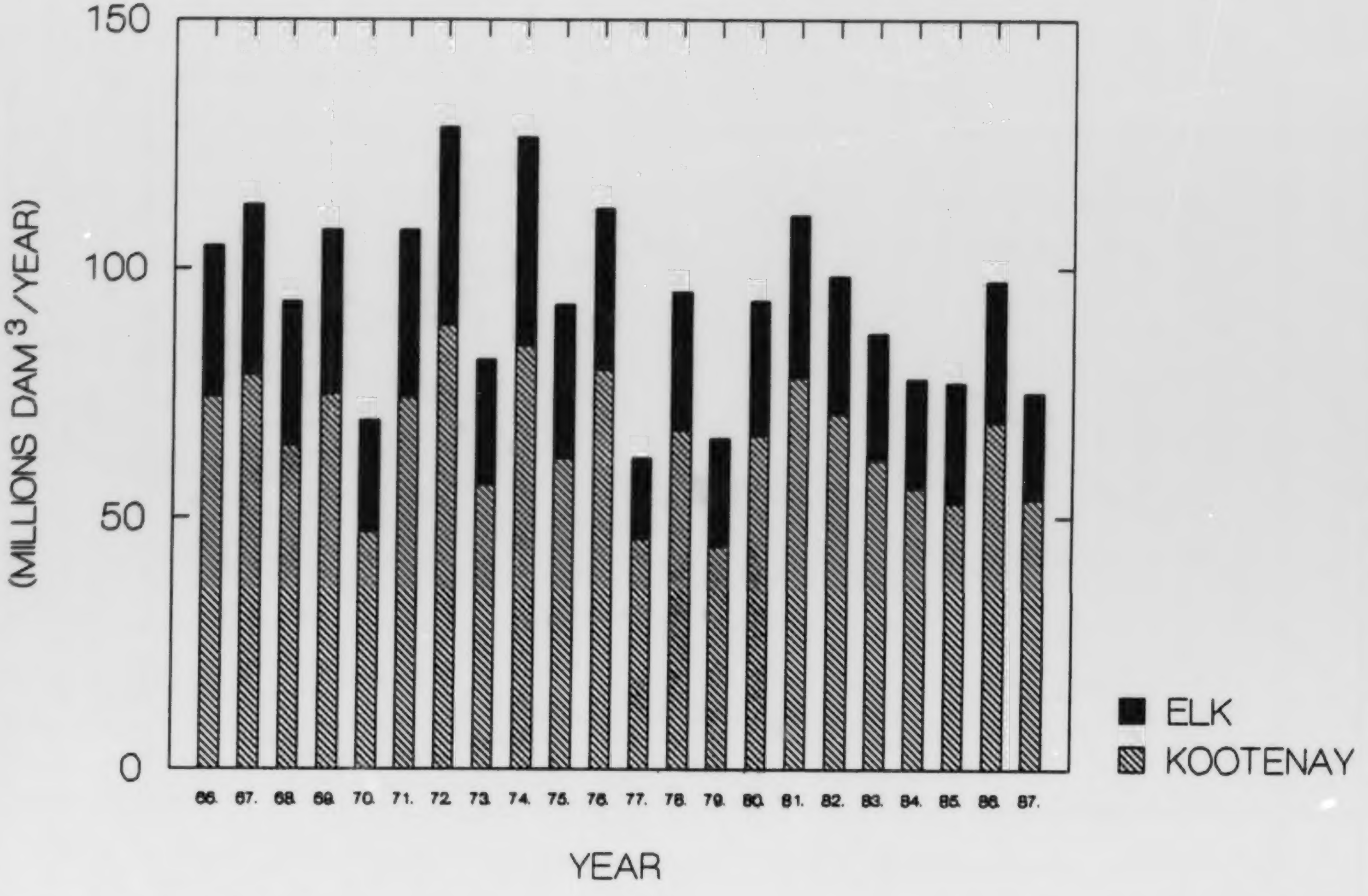


Figure 3.1.1

TOTAL ANNUAL INFLOW
OF MAJOR TRIBUTARIES

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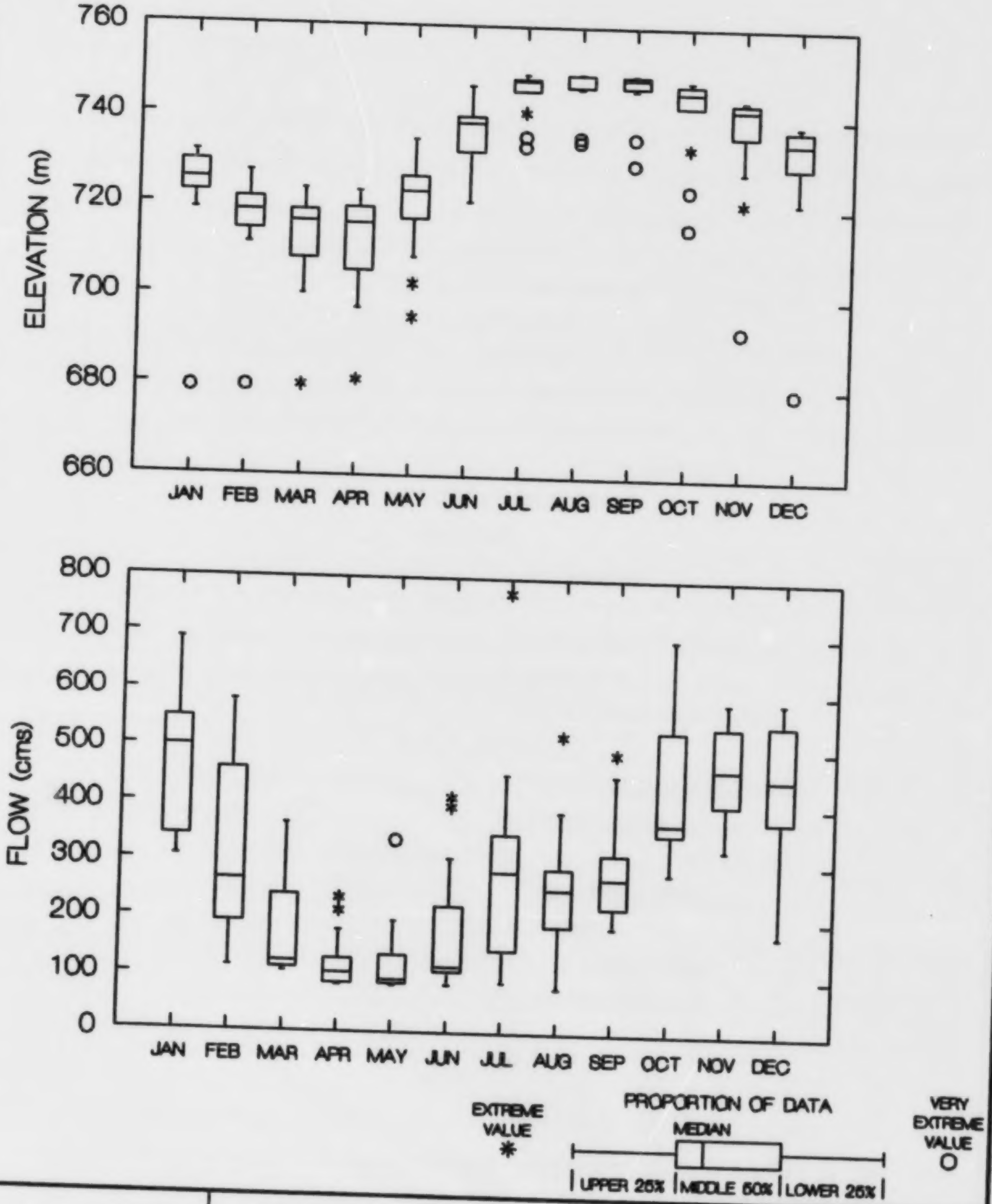

Figure 3.1.2

MONTHLY RESERVOIR RELEASE AND LAKE ELEVATION

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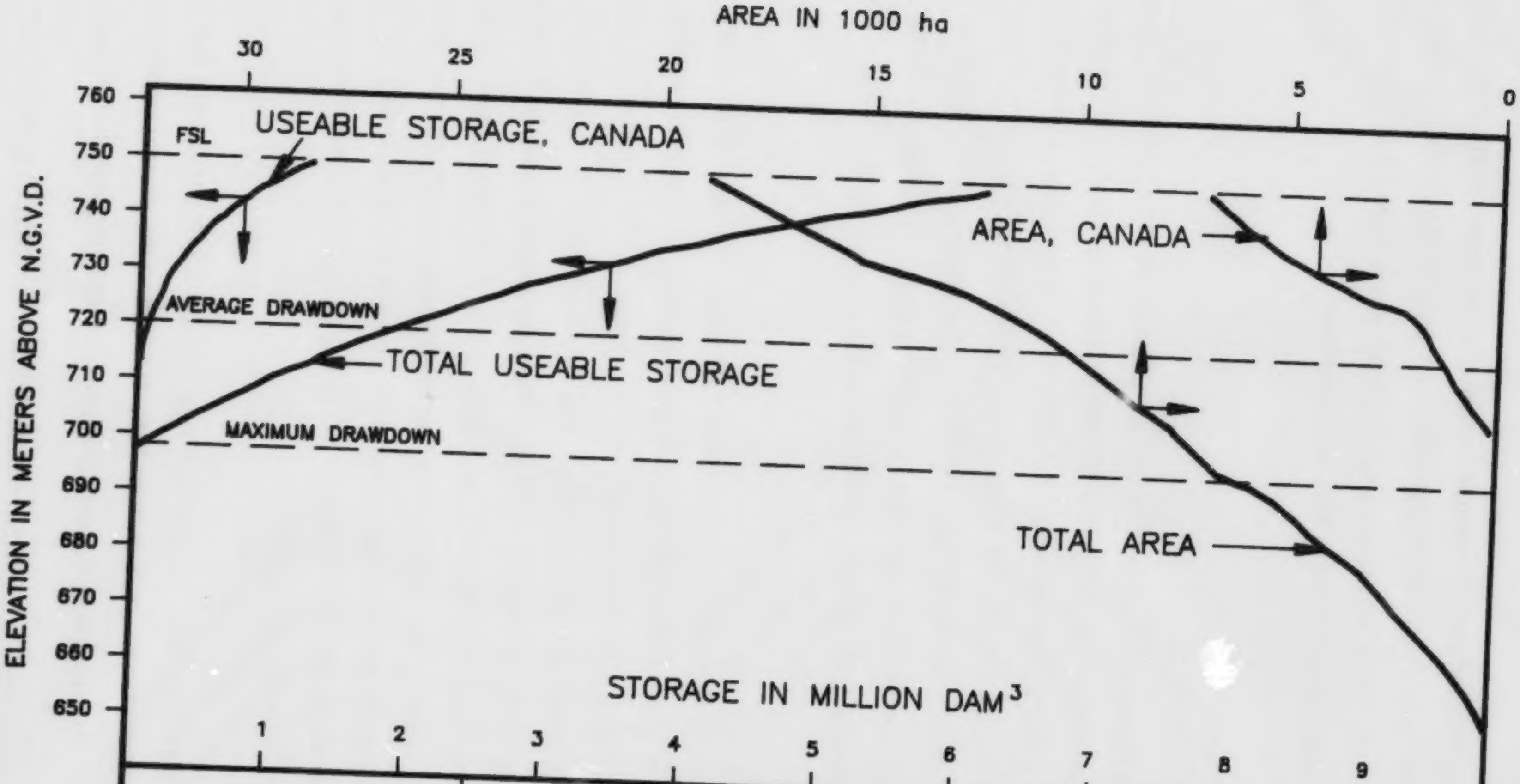

Figure 3.1.3

LIBBY DAM – LAKE KOOCANUSA

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been regulated so that temperature of the outflow approximates historical temperatures in the river downstream (between 7 and 14°C from May to September; Appendix A6, from Chisholm et al., 1989) and withdrawals in these years have been generally 18 to 21 m below surface elevations from May to November.

3.2 Temperature

Lake Koocanusa undergoes weak thermal stratification at the Border site during the summer months (Figure 3.2.1). Temperatures decrease gradually with depth in June to <10°C at 40 m. Temperatures in the surface 20 m continue to rise in July and reach a maximum temperature of about 20°C. Thermoclines form temporarily during periods of incomplete mixing in the upper 20 m of the water column, but these seldom persist for more than a few weeks and are rarely steep. Typically, the water column becomes isothermal by October or November, which coincides with the commencement of drawdown.

Since water depth increases in the reservoir increases toward the dam, other areas, away from the Border site, will have slightly different vertical temperature profiles depending on their proximity to Libby Dam. For instance, the station at Forebay, which is located 11 km upstream of the dam, has a maximum depth in excess of 100 m; the corresponding temperature profile shows a more persistent metalimnion with a steeper thermocline from June to September (Figure 3.2.2; most years from 1972 to 1978; Storm et al., 1982). The seasonal temperature maximum of approximately 20°C is reached in August, and the epilimnion layer extends to 15 m below the surface in October. Temperatures in the bottom 50 m do not vary a great deal from June to October, and fluctuate about 4°C.

By comparison, the station established near the Elk River confluence has a relatively shallow maximum depth (<35 m from 1983-1987), and the corresponding temperature profile has no thermocline and a very gradual temperature gradient; however, maximum

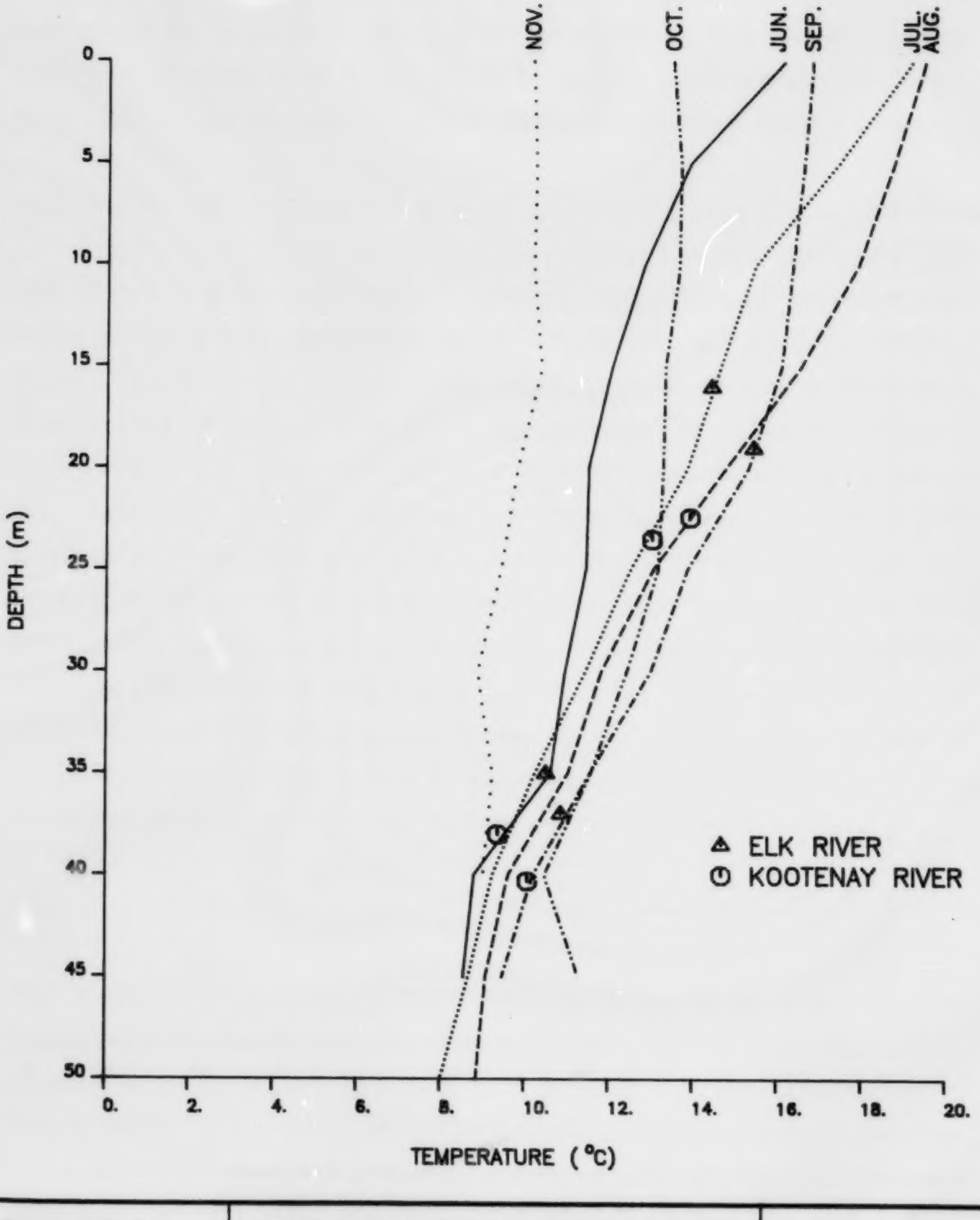

Figure 3.2.1 MONTHLY AVERAGE TEMPERATURE PROFILES AT BORDER

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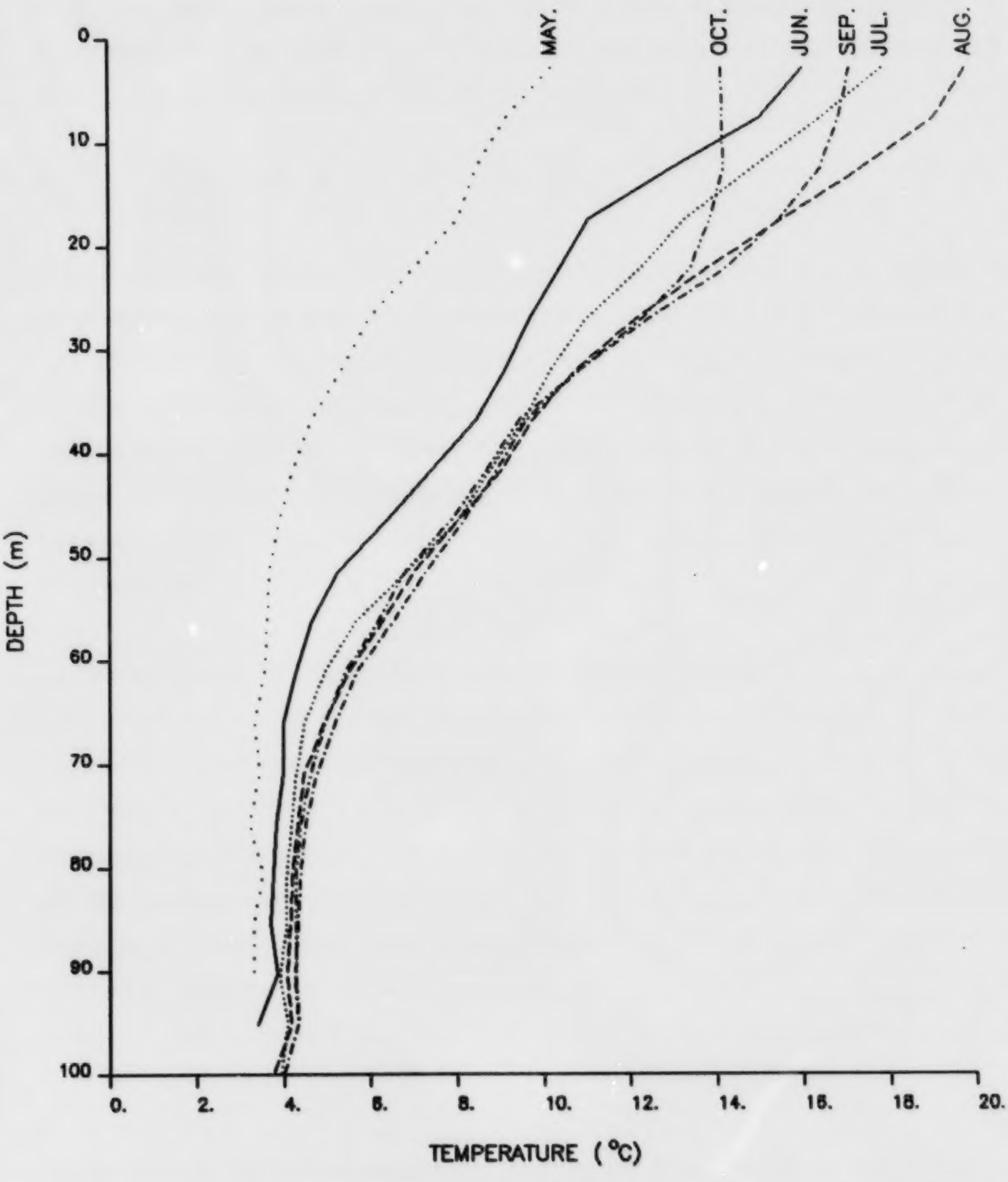

Figure 3.2.2

MONTHLY AVERAGE TEMPERATURE PROFILES AT FOREBAY

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surface temperatures are higher (approximately 24°C) and temperatures near the sediment tend to be above 10°C (Chisholm et al., 1989). The tendency for the water column to remain unstratified reflects the continual movement of water due to reservoir operations. The basin morphometry and its exposure to strong prevailing winds have also helped to keep the upper strata well-mixed throughout most of the summer.

When a river flows into a lake with a temperature gradient, the inflow seeks a depth in the water column of equal temperature (i.e. density). This phenomenon has been referred to as interflow when it occurs within the water column, or underflow when inflows are colder than any temperature within the water column and flow along the bottom. To illustrate this process, stream temperatures are indicated on temperature profiles of the Border site (Figure 3.2.1). At the Border site, spring inflow enters at approximately 35 m below the surface, and would be mixed throughout the column, since there is no thermocline. By comparison, the inflow would be expected to enter at mid depths (between 18 and 25 m) in July and August, because of increased river temperatures at this time. In the fall, underflow may occur because of colder inflow temperatures, but reservoir surface elevation would be declining rapidly at this time. Generally, inflow temperatures from the Elk and Kootenay Rivers correspond best with temperatures at mid to bottom depths of Lake Koocanusa and this is consistent with findings from Woods (1979) and Crozier and Nordin (unpublished) that the major flow appears to be along the floor of the reservoir most of the year.

3.3 Dissolved Oxygen (DO)

Since the water column does not stratify, water at the Border site tends to be well-oxygenated during most of the open-water season (Figure 3.3.1). In an average year, DO stays above 7.0 mg/L in the surface 25 m, but declines to levels approaching 5.0 mg/L below 40 m in September and October, presumably because decomposing algal material has been accumulating near the bottom throughout the summer. There is a progressive reduction in DO from June to September (>9.0 to <7.0 mg/L). At fall

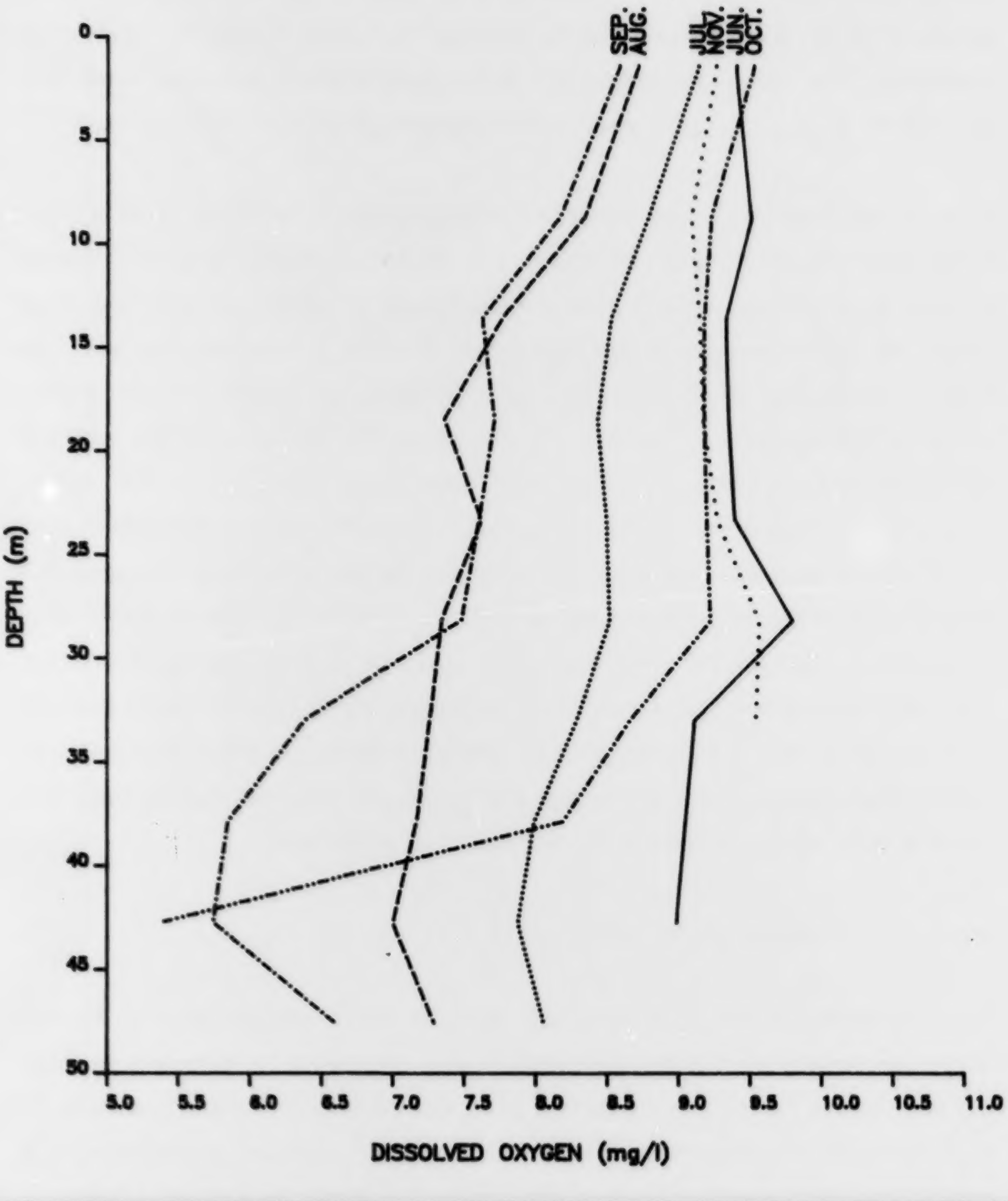

Figure 3.3.1 MONTHLY AVERAGE DISSOLVED OXYGEN PROFILES AT BORDER

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overturn, DO is replenished in the surface 30 m, although concentrations near the bottom remain low (<6 mg/L below 40 m) until November, when drawdown commences.

There are year-to-year variations with respect to the timing of fall overturn and the extent of the September oxygen depletion in the bottom waters. From 1975 to 1978, DO depth profiles indicated that the entire column was well-oxygenated (>7.0 mg/L) by September. This contrasts the situation in more recent years (1985-1989) when DO concentrations below 30 m have become substantially reduced to levels approaching 4.0 mg/L during September and October.

Further downstream, at the Forebay site, DO stays above 7.0 mg/L from June to November (Figure 3.3.2). Similar to the Border site, concentrations decline progressively from June to September throughout the water column; however, the rates of depletion are greater in both surface and bottom waters. The summer depletion of DO concentrations in the surface 30 m probably reflects the lower saturation limit of warmer water. A similar elevated depletion rate below 50 m depth (Figure 3.3.2) results from decomposition of sedimented organic matter. The interplay of these two effects results in a distinct DO maximum around 45 m depth, where water temperatures remain low (7.5°C or less) and hypolimnetic oxygen demand is not yet exerted. There is no evidence of turnover in October or November, since concentrations continue to decline at all depths during these months.

3.4 Light Penetration

Light attenuates predictably in the water column as a function of various biotic and abiotic substances that scatter and absorb light (Wetzel, 1975). A convenient measure of light attenuation is the extinction coefficient, which is influenced not only by the water itself, but by the absorbing capacity of the living particles such as plankton, and the non-living particles such as organic or inorganic sediments. A high extinction coefficient

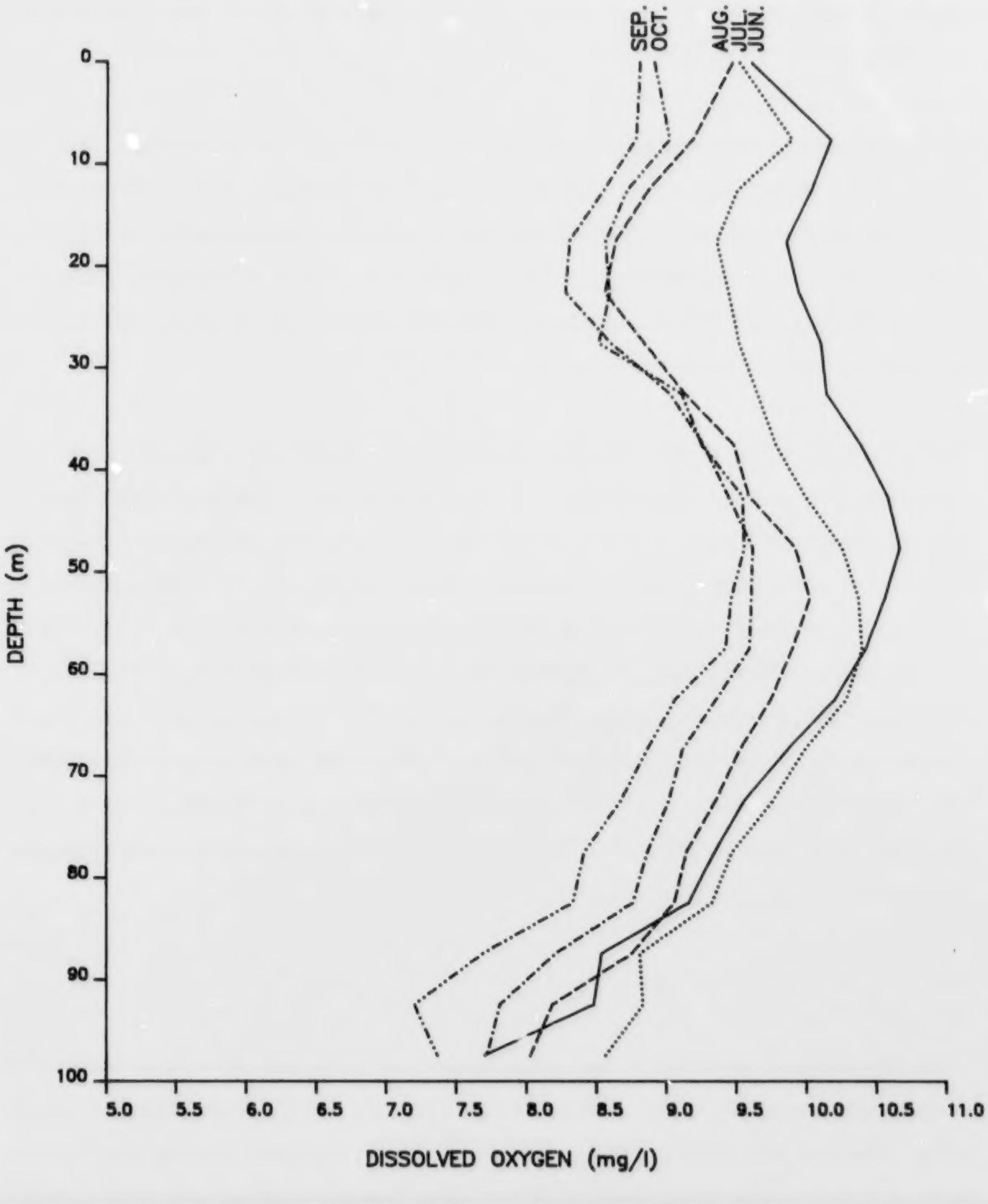

Figure 3.3.2

MONTHLY AVERAGE DISSOLVED OXYGEN PROFILES AT FOREBAY

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(>2.0) may be a result of either high primary production (i.e. lake is eutrophic), high turbidity from spring runoff, or dystrophy, a staining of the lake water by humic acids. Lake Koocanusa has a relatively high mean extinction coefficient of 2.77 (Woods, 1979), which primarily reflects high sediment loading from the tributaries during spring runoff (Bonde and Bush, 1975; Rocchini, 1981; Storm et al., 1982).

Water transparency is another useful measure of the quality of light in lakes. An approximate means of measuring this transparency is to observe the depth at which the Secchi disk (20 cm in diameter in this study) disappears from view as it is lowered into the water column. There was no significant long-term trend in Secchi depth in Koocanusa Reservoir, indicating that the light penetration has not changed significantly over the past 17 years.

Many studies have shown that light scattering by particulate matter has a disproportionately greater influence on Secchi depth measurements than does dissolved organic matter (Wetzel, 1975). In Lake Koocanusa, both turbidity and non-filterable residue (NFR) are significantly correlated with Secchi depth (Table 3.4.1) which confirms that light penetration in the reservoir is greatly influenced by presence of suspended solids. Secchi depth was also negatively correlated with total organic carbon and phytoplankton biomass, both of which are measures of particulate organic matter that would scatter light in the reservoir; surprisingly, however, Secchi depth was not significantly correlated with chlorophyll a (Table 3.4.1) even though it is an index of phytoplankton biomass.

Chlorophyll a content varies widely among species of algae, and this factor may have been sufficient to remove any correlation with Secchi depth. This seems anomalous given the significant correlations with total organic carbon and phytoplankton biomass, but these correlations are not strong (that with TOC accounts for 4% of the variation and that with phytoplankton, 7%) and are based on larger sample sizes.

Table 3.4.1 Correlations Between Secchi Depth and Variables Affecting Water Clarity
* indicates significant correlation at alpha = 0.05

Variable	Pearson Correlation with Secchi Depth	N
Turbidity	−0.293*	129
NFR	−0.243*	104
Total Organic Carbon	−0.190*	121
Chlorophyll a	0.024	37
Phytoplankton Biomass	−0.272*	57
Zooplankton Biomass	0.053	63

In an average year, Secchi depth increases from 0.5 m in May to >5 m in September, and decreases slightly in October and November as drawdown commences (Figure 3.4.1). Secchi depth values averaged over the entire reservoir in 1986 showed a similar trend, with highest values in August (5.8 m) and lowest values in May and June (1-2.3 m) (Chisholm et al., 1989). Approximately the reverse seasonal trend occurs for NFR (Figure 3.4.1); median NFR concentrations decline from a May measurement of 14 mg/L to an August low of <4 mg/L. These measurements are more variable than Secchi depths (especially in spring), reflecting the year-to-year variation in timing of the spring freshet and reservoir filling. There have been no significant changes in NFR levels over the past 17 years at the Border site, and there are no data available for the Forebay site to permit comment on seasonal or long-term trends.

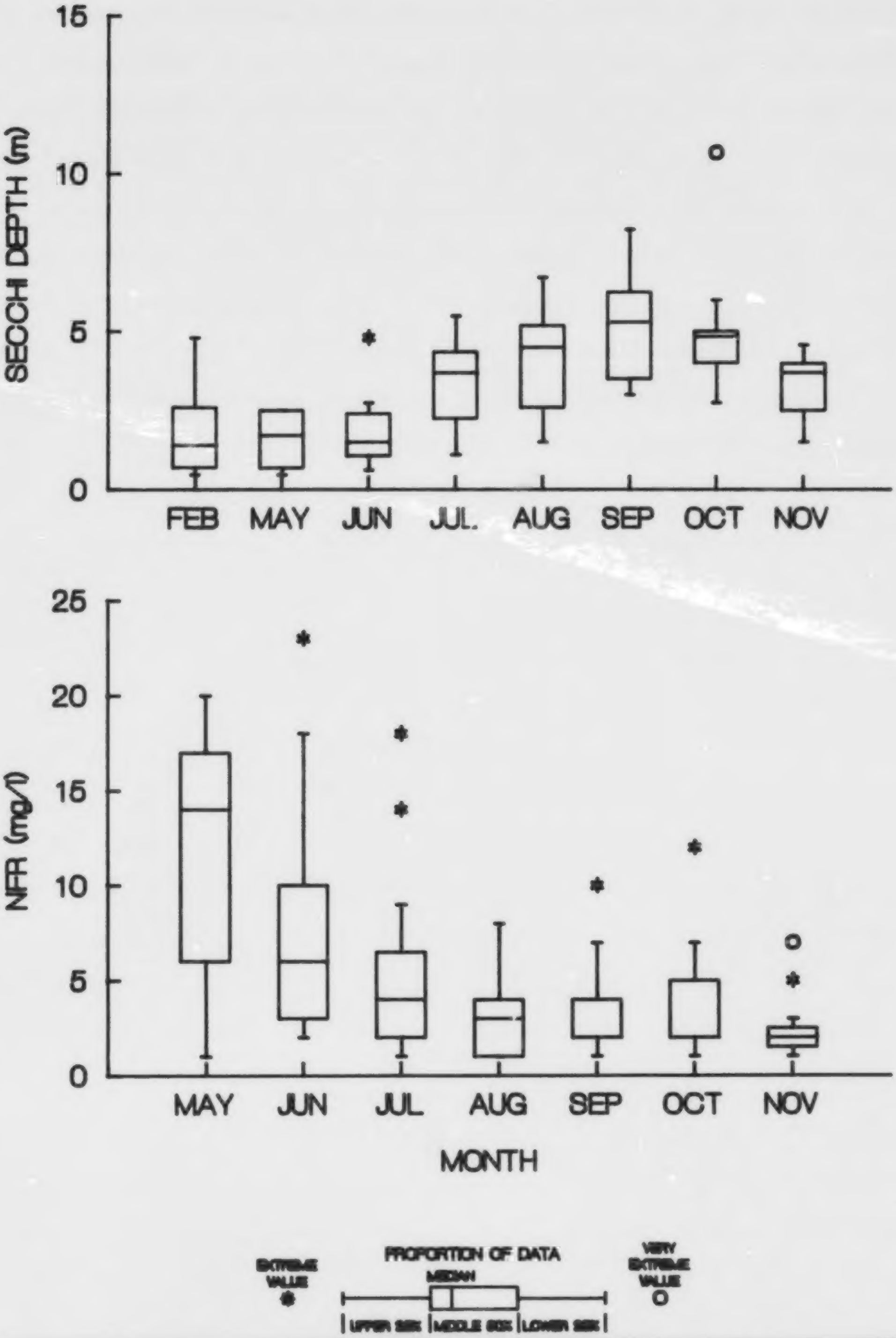

Figure 3.4.1

BORDER SITE
WATER CLARITY & SUSPENDED SOLIDS

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4.0 TRIBUTARY CHEMISTRY

4.1 Metals

Long-term trends in concentrations of metals are summarized in Table 4.1.1. Time-series analysis of metals data was complicated by frequent changes in analytical methods over the years, and a steady improvement in sensitivity and detection limits. Thus, statistics alone are not always sufficient to separate effects of changes in waste discharges or other interventions in the basin from apparent changes arising from better methods.

Plots of yearly metals concentrations (Figure 4.1.1) reveal a drop in levels of lead and zinc, and to a lesser extent iron and manganese, after 1979. For iron and zinc, there was a significant long-term trend of declining concentrations through time (Table 4.1.1). These changes may be directly attributed to the commissioning, in October 1979, of Cominco's drainage water treatment plant, which treats acidic drainage water from the Sullivan mine and waste water from the concentrator tailings ponds (L. McDonald, personal communication). Mean concentrations of metals in mine water, discharged directly to a tributary of the Kootenay River before 1979 were: total Fe, 345 mg/L; dissolved Mn, 34 mg/L; total Pb, 6.8 mg/L; total Zn, 102 mg/L (Rocchini, 1981). In contrast, typical levels for all forms of these same metals in effluent from the treatment plant today are: Fe, 2.5 mg/L; Mn, 0.25 mg/L; Pb, 0.030 mg/L; Zn, 0.25 mg/L (L. McDonald, personal communication).

In the plot of total zinc concentrations (Figure 4.1.1), an abrupt decline is evident between 1979 and 1980, which corresponds exactly to the implementation of wastewater treatment at Cominco Kimberley Operations. In fact, since analysis of a subset of the data indicates no significant trend from 1980-1985 (Table 4.1.1), it must be the drop from 1979-1980 that is responsible for the overall trend of declining concentrations. The same interpretation would apply to the statistical trend in iron

Table 4.1.1 Results of Time Series Analysis of Total Metal Concentrations in the Tributaries. NS = no statistically significant trend.

Site	Variable	Time Period Tested	Significant Trend*	Slope**
Kootenay R.	Total Iron	1973/1 - 1989/4	decrease	-0.03667
	Total Copper	1973/1 - 1986/5	NS	
	Total Manganese	1983/5 - 1989/4	NS	
	Total Zinc	1973/1 - 1989/5	decrease	-0.00256
	Total Lead	1979/8 - 1989/5	NS	
Elk R.	Total Iron	1982/7 - 1989/4	decrease	-0.00929
	Total Copper	1977/6 - 1986/5	NS	
	Total Manganese	1983/8 - 1989/4	NS	
	Total Zinc	1986/9 - 1989/5	NS	
		1979/1 - 1988/3	NS	
	Total Lead	1978/1 - 1989/5	NS	

* as indicated by a Seasonal Kendall Tau test statistic at alpha = 0.05

** slope indicated the rate of change in mg/L per year

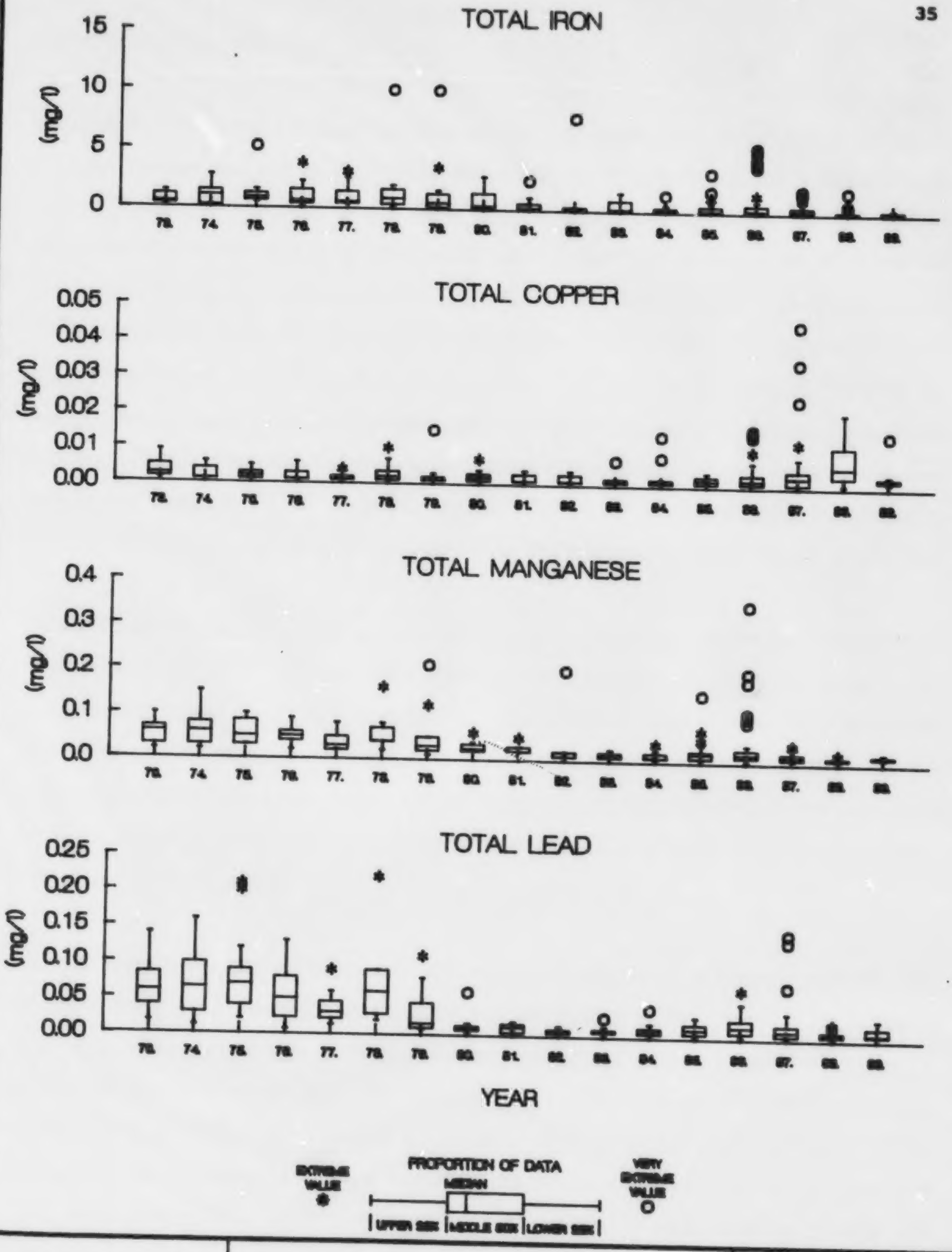
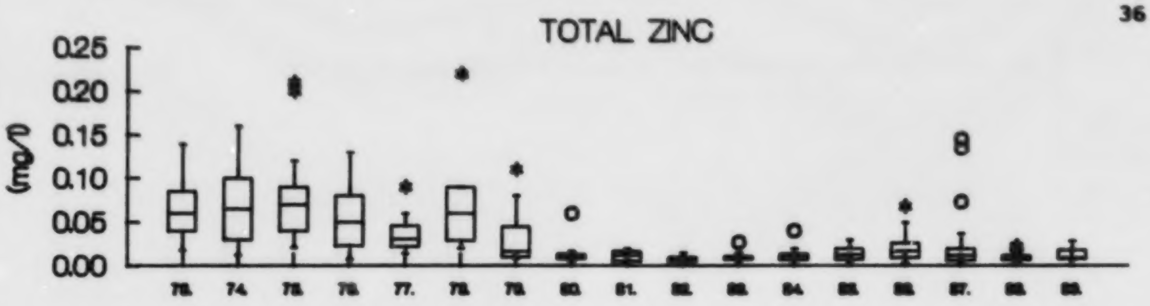


Figure 4.1.1	KOOTENAY RIVER TOTAL METALS	HydroQual Canada Ltd


TOTAL ZINC

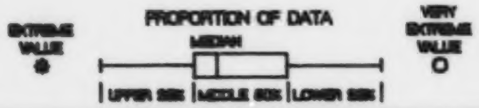

EXTREME VALUE
PROPORTION OF DATA
MEDIAN
UPPER 25%
MIDDLE 50%
LOWER 25%
VERY EXTREME VALUE

Figure 4.1.1
(continued)

KOOTENAY RIVER
TOTAL METALS

HydroQual
Canada Ltd.

concentrations as well.

Inspection of Figure 4.1.1 suggests that an identical pattern to zinc and iron occurred for total manganese and lead. A time-series test for long-term trends in these metals was not possible because of improvements in analytical sensitivity and numerous detection limit revisions through the years. It was possible, however, to do a median test comparing concentrations before and after 1979. The median lead concentration declined by 0.05 mg/L, and the median manganese concentration by 0.025 mg/L, after 1979 (Table 4.1.1). This represents a decrease of 30-40% for lead and >70% for manganese. Hence, initiation of wastewater treatment at Cominco Kimberley Operations was immediately reflected in sharply lower concentrations of iron, zinc, manganese, and lead in the Kootenay River. As is evident from Figure 4.1.1, annual variability of metals concentrations was also sharply reduced after the treatment plant was commissioned.

Data for total copper do not share the conspicuous drop in concentration observed for other metals (Figure 4.1.1). Data from 1978 to 1986 revealed no significant linear trend. Copper has never been released in high concentrations from Cominco Kimberley Operations. The mean concentration in Sullivan mine water from 1975-1978 was 0.3 mg/L total copper (Rocchini, 1981), while in recent years copper levels in the discharge from the drainage water treatment plant have remained near 0.03 mg/L (L. McDonald, personal communication).

Copper data from mid-1986 onward are suspect because of a known problem with contamination from incorrect washing of sample bottles (Clark, 1990). As a result, median copper concentrations in recent years are slightly elevated, however values for some samples are extremely high, resulting in an unusual number of extreme values in the box-and-whisker plots (Figure 4.1.1). The same pattern of more frequent outliers occurs for all five metals examined, and at about the same time (Figure 4.1.1). For total zinc, this resulted in a significant linear trend of increasing concentration if 1980-1989 data are considered (slope = 5×10^{-4} mg L^{-1} yr^{-1}), although as previously mentioned, 1980-1985 data show no trend. This similarity suggests that the contamination problem was pervasive for all five metals; data for 1988 and 1989, analyzed by the Canada Centre for Inland Waters, show typical low values again.

Concentrations of most metals, and especially zinc and lead, are consistently lower in the Elk River than in the Kootenay River (compare Figures 4.1.1 and 4.1.2). The Elk River valley lacks the sulphide ore deposits found in the Purcell Mountains on the west side of the Kootenay River basin. There are no metal mines in the Elk River valley, and the coal, mined at five large surface mines (Figure 1.1) is low in sulphur and produces no acid drainage water. Trend analysis did reveal a decrease in iron concentrations through time in the Elk River, although the slope was extremely low (-0.0093 mg L^{-1} yr^{-1} and probably an artifact of increasing analytical precision. As would be expected, none of the other metals showed a significant temporal trend in concentrations (Table 4.1.1). Levels of total zinc appear to be gradually increasing (Figure 4.1.2) but the trend is not statistically significant, either for the long term (1979-1988) or in the more recent data (1986-1989). Levels of copper and manganese in the Elk River remained so low that until 1978 they could not be detected, and lead concentrations have remained near zero throughout the sampling period (Figure 4.1.2). Finally, as for the Kootenay River, there are numerous extreme values for all metals except lead in 1986-1988. This is almost certainly a contamination problem, as discussed above.

4.2 Nutrients

Kootenay River

Table 4.2.1 presents results of time-series analysis for nutrients. The first five columns summarize the seasonal Kendall's Tau test for long-term trends (including the slope value and its direction, if significant), while the last two columns summarize median tests comparing differences between two distinct periods.

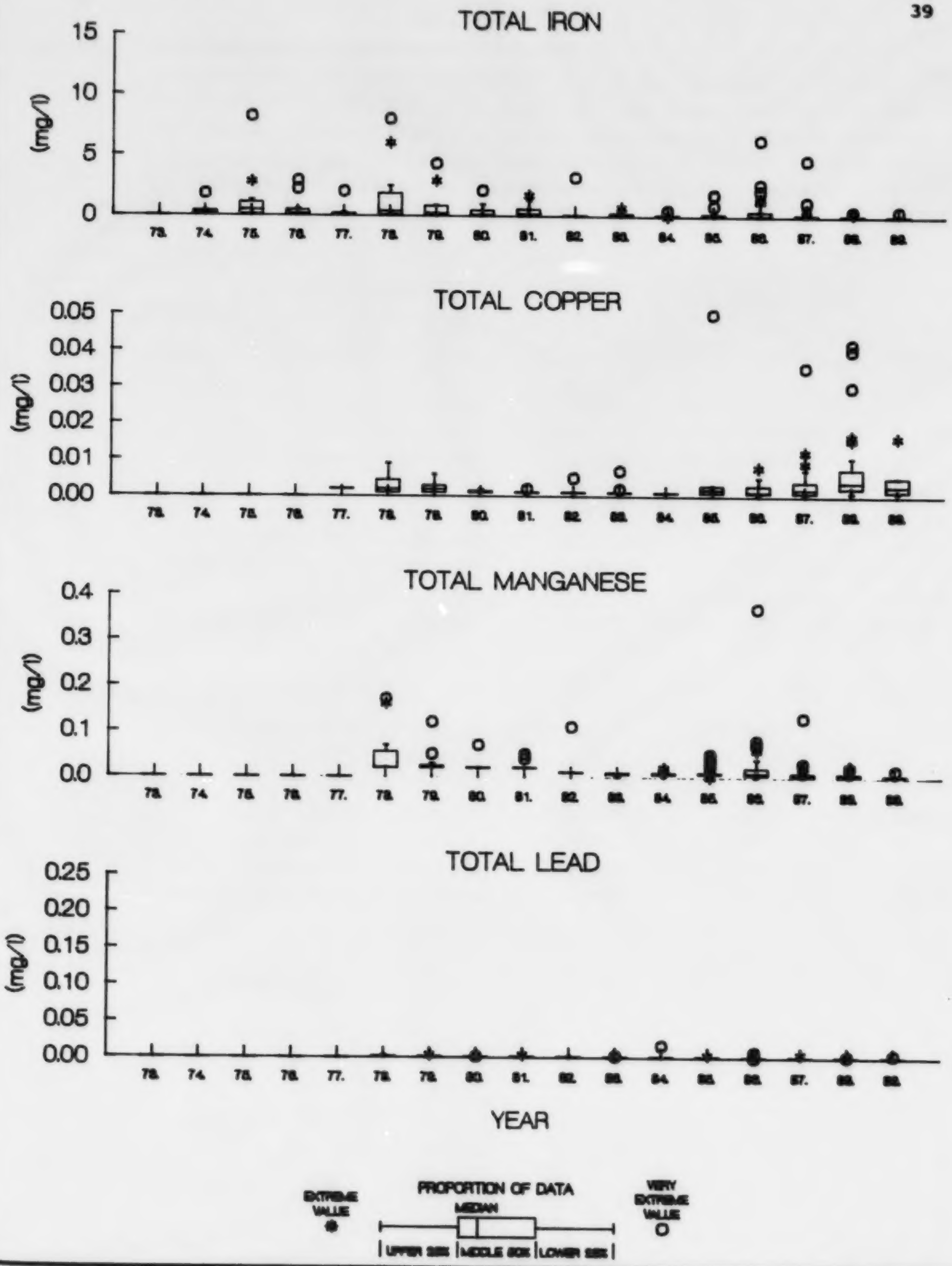
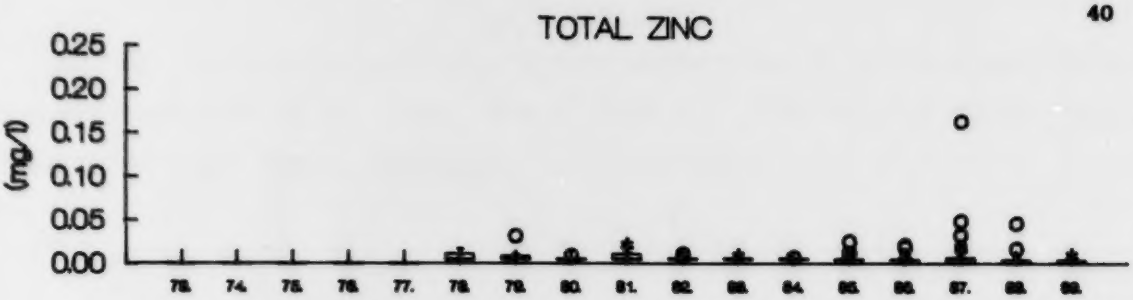


Figure 4.1.2	ELK RIVER TOTAL METALS	HydroQual Canada Ltd.



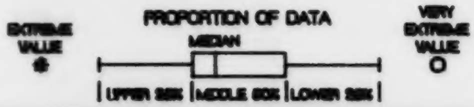

EXTREME VALUE
PROPORTION OF DATA
MEDIAN
VERY EXTREME VALUE
UPPER 25%
MIDDLE 50%
LOWER 25%

Figure 4.1.2 (continued)	ELK RIVER TOTAL METALS	HydroQual Canada Ltd.

Table 4.2.1 **Time Series Analyses of Nutrients in Tributaries.**

Long–term trends in nutrient concentrations as indicated by a Seasonal Kendall Tau Test (alpha = 0.05); NS = Not Statistically Significant; Slope refers to the rate of change mg/L per year. "Median" refers to a Wilcoxin signed rank test comparing medians between two adjacent time periods; "Date" refers to the time period that were compared and "Difference" refers to the decrease or increase noted. N/A = Not Applicable

Site	Variable	Seasonal Kendall Tau Period	Trend	Slope	Median Test Periods	Difference (mg/L)
Kootenay	TP	1976–1989	decrease	–0.002	1971/6–1975/12 1976/6–1980/12	–0.181
	OP	1976–1989	decrease	–0.0004	1971/6–1975/12 1976/6–1980/12	–0.058
	TN	1976–1989	NS	N/A	N/A	N/A
	TKN	1974–1989	decrease	–0.0035	N/A	N/A
	NH3	1977–1988	decrease	–0.0004	1971/7–1976/12 1977/7–1981/12	–0.035
	N03+N02	1974–1988	increase	0.0055	N/A	N/A
Elk	TP	1971–1989	NS	N/A	N/A	N/A
	OP	1972–1989	decrease	–0.0001	1969/6–1972/4 1972/6–1975/4	–0.003
	TN	1974–1989	increase	0.0055	N/A	N/A
	TKN	1974–1989	NS	N/A	N/A	N/A
	NH3	1974–1989	decrease	–0.0003	N/A	N/A
	NO3+NO2	1975–1989	increase	0.0057	N/A	N/A

Since 1976, annual average TP and SRP concentrations have declined significantly in the Kootenay River (measured below the confluence of the St. Mary River) (Table 4.2.1). The biggest single reduction occurred during 1975, when Cominco's fertilizer operations commenced recycling phosphorus-rich supernatant from the Gypsum pond as cooling water rather than discharging it to the river (Figures 4.2.1 and 4.2.2). Corresponding median tests indicated that TP and SRP concentrations for the five-year period ending in 1975 were an average of 0.181 mg/L (TP) and 0.058 mg/L (SRP greater than for an equal five-year period after 1975 (Table 4.2.1). A series of other improvements in wastewater treatment were made between 1975 and 1987, when the fertilizer plant finally shut down, and these further contributed to the reduction in phosphorus concentrations.

The persistently low TP concentrations have not changed significantly over the past 18 years in the Elk River (Figure 4.2.1; Table 4.2.1). The already low SRP concentrations in the Elk River have decreased significantly, but the magnitude of change is very slight compared with that in the Kootenay River (Table 4.2.1). This reduction is probably an artifact due to differences in analytical techniques used before and after 1972. McDonald (1987) found that the major source of dissolved phosphorus in the Elk River drainage is natural, from Michel Creek, particularly during spring runoff. Industrial and municipal sources were small by comparison.

It is evident from Figures 4.2.1 and 4.2.2 that the quantity of phosphorus entering Lake Koocanusa from Cominco's fertilizer plant operations in the early 1970's was far greater than either the background levels in the Elk River, or the present-day loads from the Kootenay River. Figure 4.2.3 shows SRP concentrations since 1976 in both tributaries on an expanded scale. The similarity in SRP concentrations between the Kootenay and Elk Rivers is evident; although the portion of this phosphorus attributable to human activity (sewage discharge, agriculture, etc.) is unknown, the similarity of temporal patterns in the basins suggest that SRP concentrations in both are presently dominated by natural sources.

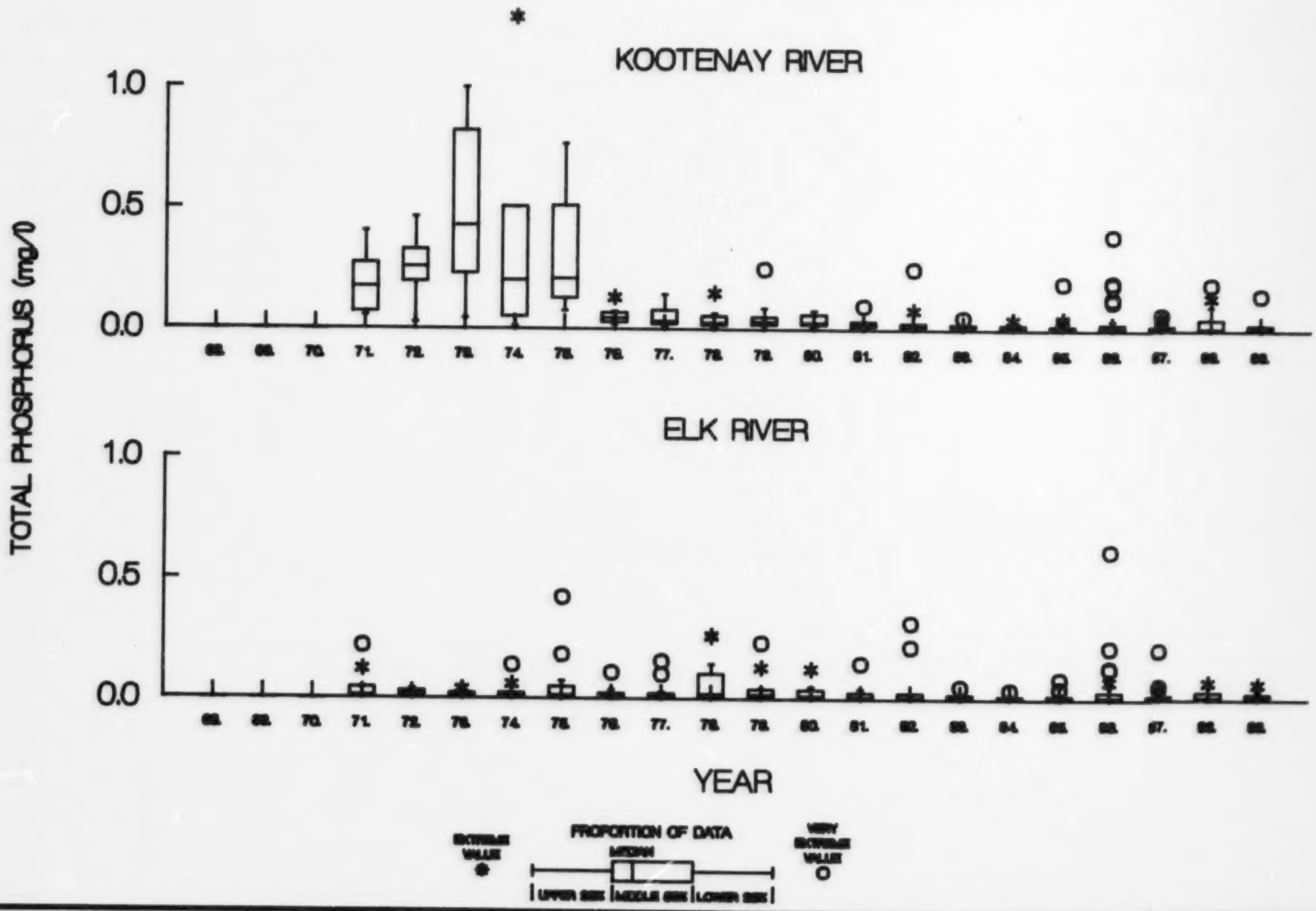


Figure 4.2.1

ANNUAL INFLOW
TOTAL PHOSPHORUS CONCENTRATIONS

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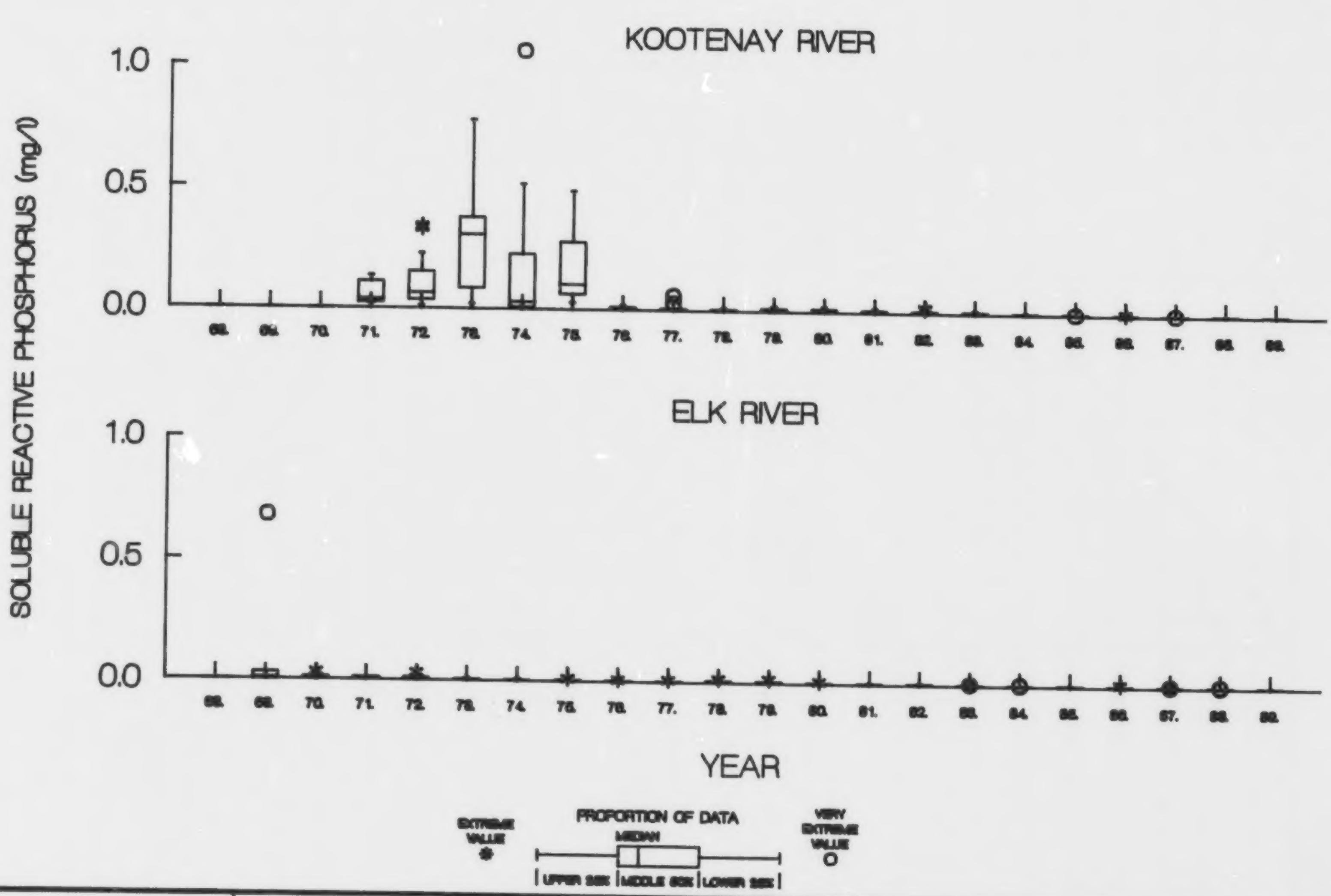

Figure 4.2.2

ANNUAL INFLOW
SOLUBLE REACTIVE PHOSPHORUS CONCENTRATIONS

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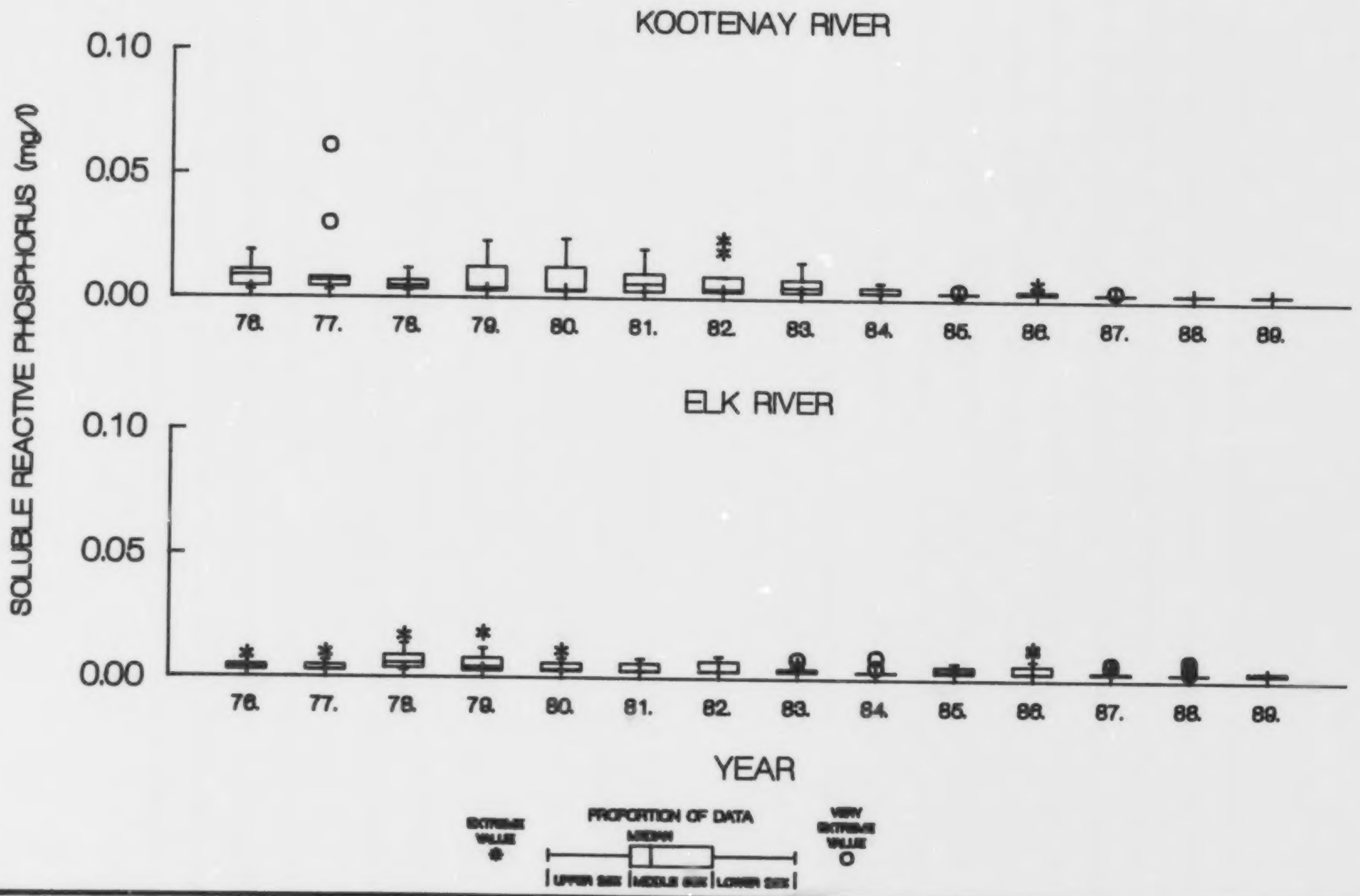

Figure 4.2.3

ANNUAL INFLOW
SOLUBLE REACTIVE PHOSPHORUS CONCENTRATIONS

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Nitrogen

There is no significant trend in concentrations of total N in the Kootenay River, (Figure 4.2.4, Table 4.2.1), but there have been changes in some of the component nitrogen forms. Concentrations of total Kjeldahl nitrogen and ammonia both declined significantly from the mid-1970's to 1988 (Table 4.2.1); over the same period there was no significant change in concentrations of nitrate plus nitrite. The decline in ammonia concentrations does not follow the steady trend found for TKN. Rather, most of the decline in ammonia occurred suddenly in 1974-1976 (Figure 4.2.5). Median NH_3 concentrations in 1971-1976 were 0.035 mg/L greater than in 1977-1981 (Table 4.2.1). The drop in ammonia concentrations in the mid-1970's resulted from improvements in waste treatment at Cominco's fertilizer plant. During 1975-78 this effluent was found to have a maximum ammonia concentration of 51.5 mg/L, creating a loading of 104 kg/D (Rocchini, 1981).

In contrast to the Kootenay River, nitrate-nitrite concentrations in the Elk River have increased consistently at an average rate of 0.0057 mg/L per year, as a result of residuals from nitrate-based explosives used in coal mining (Figure 4.2.6; McDonald, 1987). Modern surface mining started in 1971 and activity has steadily increased, particularly in the early 1980's. There was a large increase in nitrate+nitrite concentrations from 1975 to 1976, when the Fording Mine began using slurry gel explosives during wet blasting conditions (Pommen, 1983).

There has been no significant change in total Kjeldahl nitrogen from 1974 to 1989, but total nitrogen has steadily increased (Figure 4.2.4). The slope of the total nitrogen increase nearly exactly equals the slope of the increase in NO_3 + NO_2, indicating that nitrate residues from coal mining explosives are also responsible for the increase in total nitrogen.

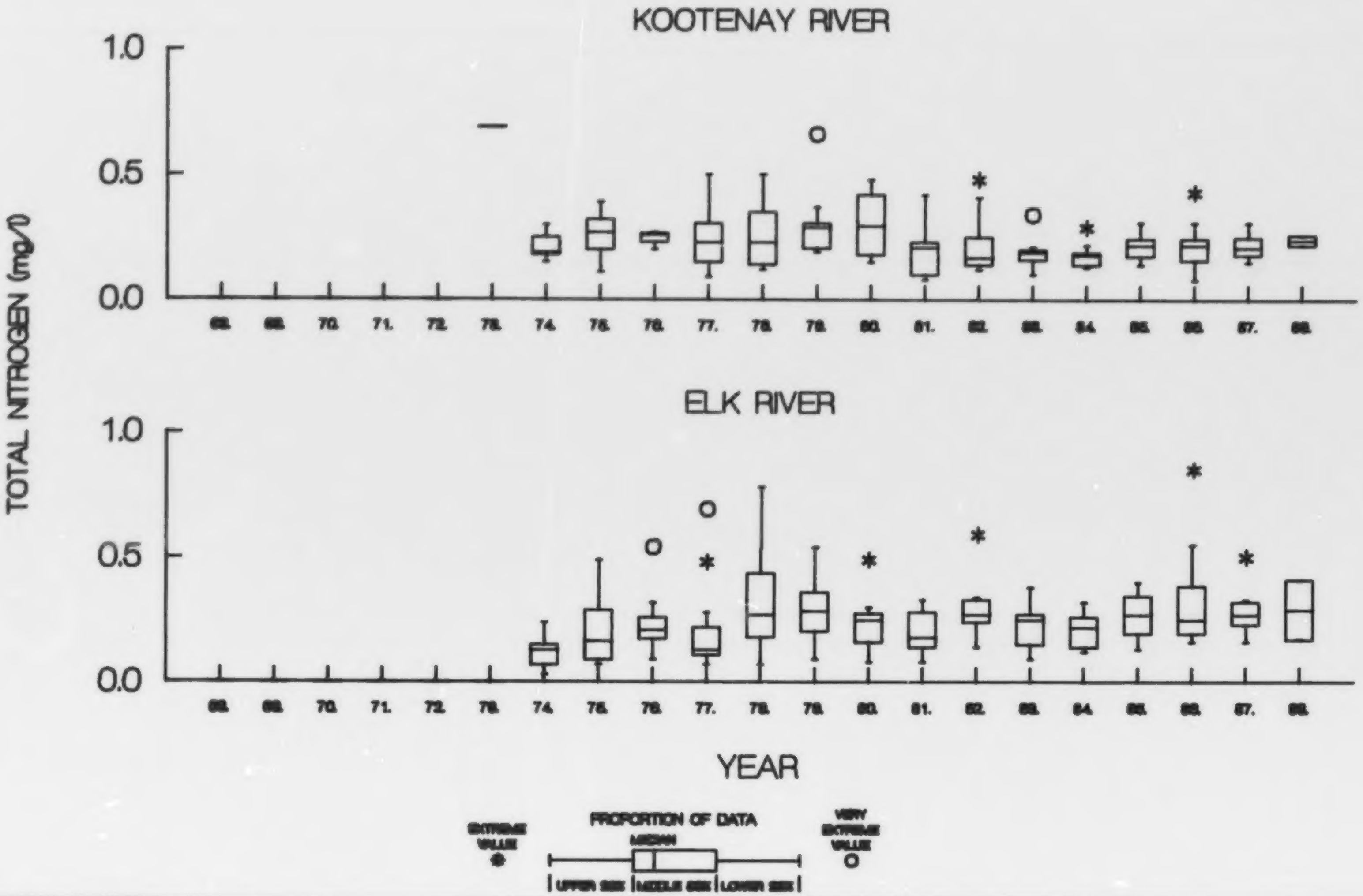

Figure 4.2.4

ANNUAL INFLOW
TOTAL NITROGEN CONCENTRATIONS

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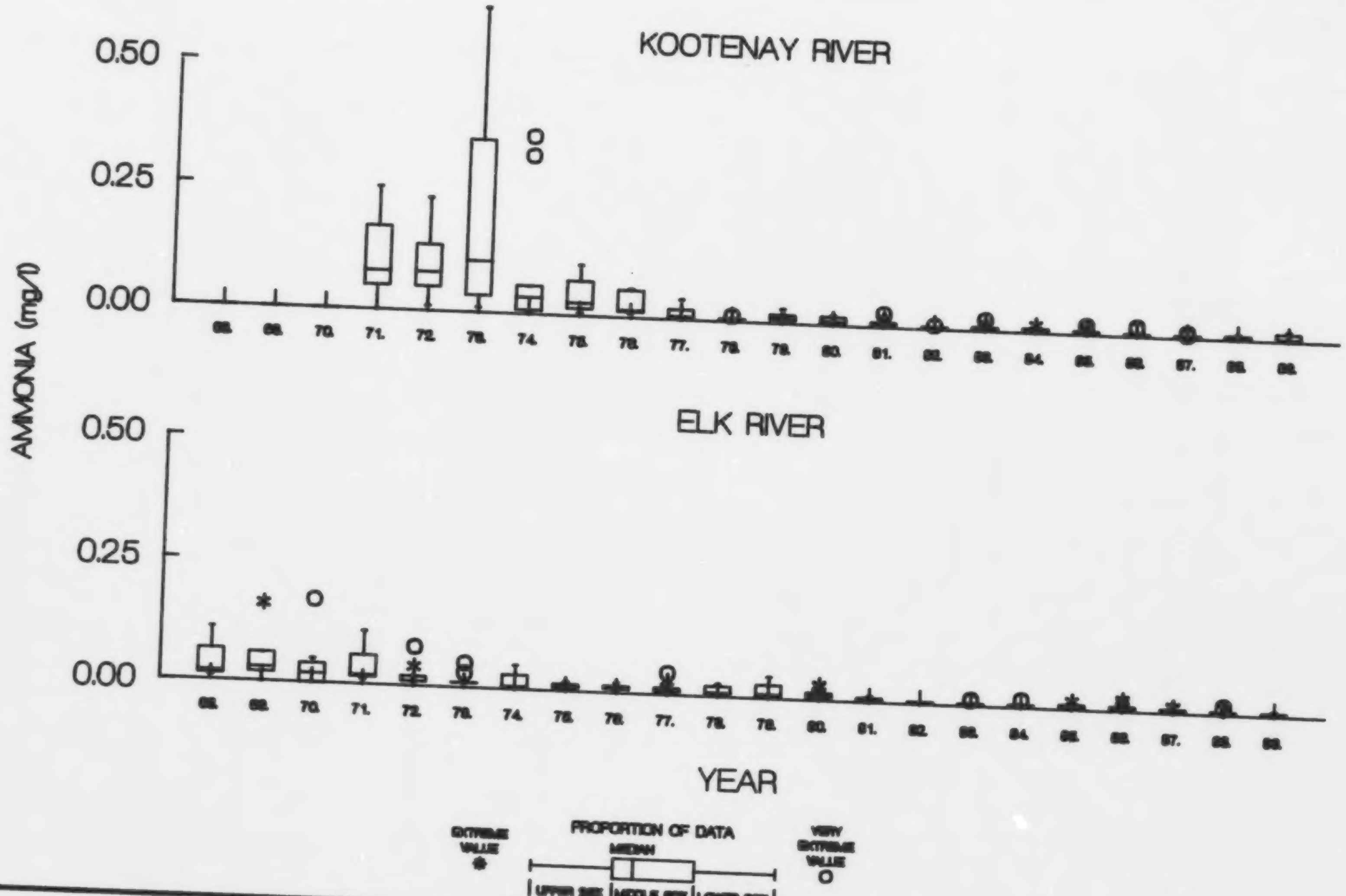

Figure 4.2.5

ANNUAL INFLOW
AMMONIA CONCENTRATIONS

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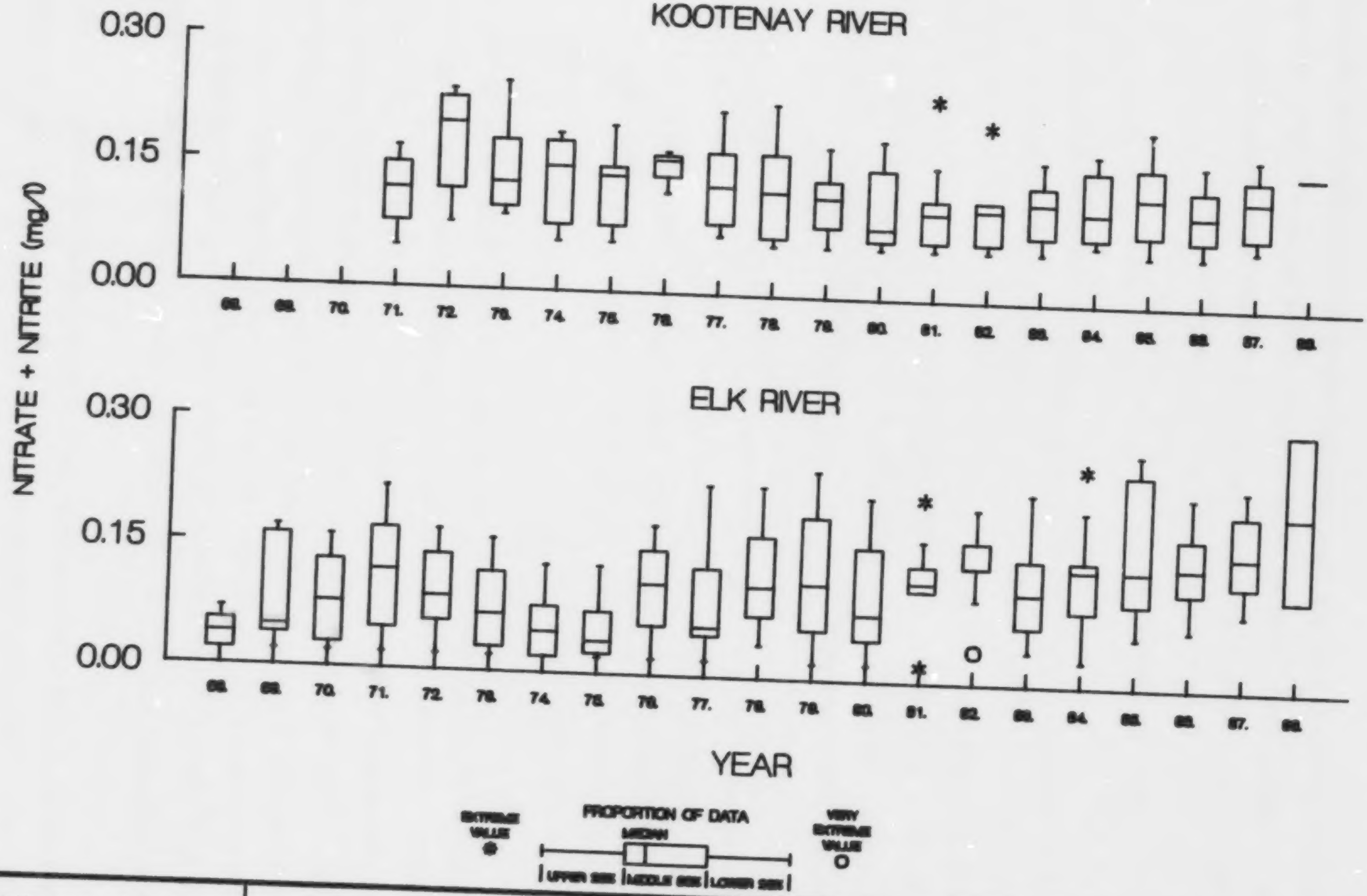

Figure 4.2.6

ANNUAL INFLOW

NITRATE + NITRITE CONCENTRATIONS

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Ammonia concentrations in the Elk River have significantly declined since 1974, although the rate is fleetingly small (Table 4.2.1). Most of the decline appears to have occurred in the 1970's. The decline is so small however, that it could be a natural fluctuation.

4.3 Other Parameters

Other variables of interest include colour, concentrations of total dissolved solids (TDS), non-filterable residue (NFR) and fluoride (F). Three different methods (true colour, TC; total absorbence colour, TAC; and single-wavelength colour, SWL) have been used to measure colour in the water. Most of the early data are total absorbence colour; more recent data are single-wavelength colour. Since most of the TAC data were generated by measuring light absorption at the same wavelength as is normally used to measure SWL colour (465 nm), these data have been pooled in Figure 4.3.1. Generally, values for the Kootenay River were higher than those for the Elk River for each of the three methods. The deeper colour of the Kootenay River reflects effluent loading from the Crestbrook Forest Industries pulp mill near Skookumchuck. The colour of kraft effluent is attributed to oxidation of tannin and lignin-like compounds and other organic wood extractives (Rocchini, 1981).

Throughout the 1970's attempts to reduce the colour effects of the Crestwood mill's effluent on the Kootenay River were largely unsuccessful, especially during low flow when dilution dips to about 1:40.5 (Rocchini, 1981). In the fall of 1981 mill effluent was diverted to a series of rapid infiltration basins, which have largely returned the Kootenay River to its natural colour (L. McDonald, personal communication). Since the system was only working intermittently from 1981 to 1982, we compared colour values between 1978 and 1980 with those between 1983 and 1985; the test confirmed that there was a significant reduction of 13.8 colour units after rapid infiltration was commenced.

There have been no significant changes in TDS for either the Kootenay or Elk River since 1971. TDS concentrations in the alkaline waters tend to be dominated by

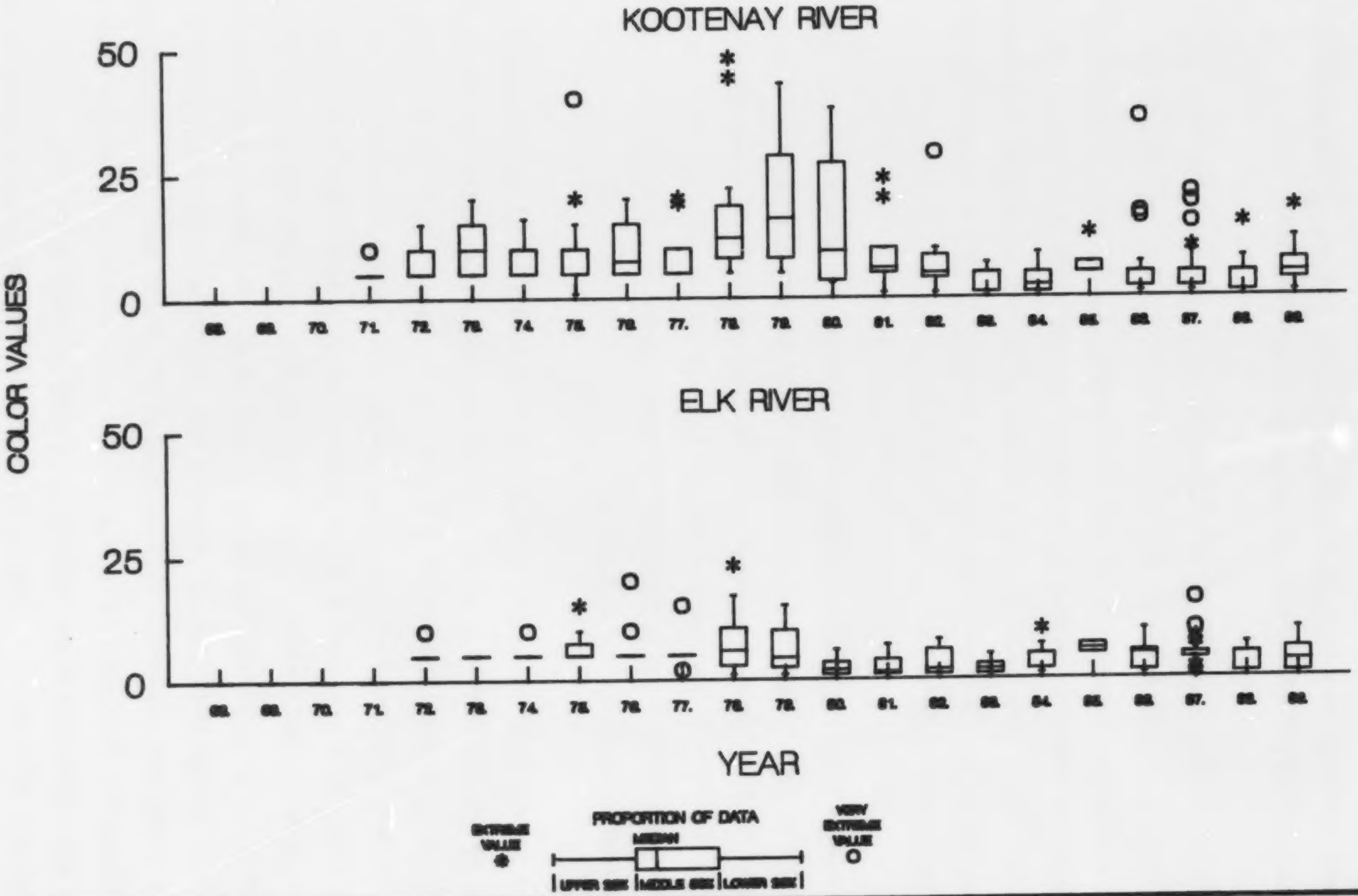

Figure 4.3.1

ANNUAL COLOR VALUES

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naturally occurring ions such as Ca, Mg and HCO_3 and are largely unaffected by anthropogenic activities of the types underway in these basins.

NFR concentrations have not changed significantly in the Kootenay River, but they have decreased significantly in the Elk River since 1972 (Table 4.3.1). The data show a drop in concentrations between 1982 and 1983 (Figure 4.3.2), so a median test was performed to measure the magnitude of the change. The test measured a small decrease of 2.75 mg/L (Table 4.3.1) almost exactly the decline predicted over the entire 18 year period by the slope of the trend line (2.5 mg/L). Hence, most of the decline over the monitored period appears to result from a drop in 1982.

Compared with normal variability of NFR in the Elk River, the observed decline is small. While it may reflect improved forestry practices in the basin, or control of sediment release from coal mines, it could as easily have been caused by consecutive, recent years of low flow in the river. A much more detailed and specific NFR sampling program would be needed to assess the real effects of watershed land use on suspended solids levels in the Elk River.

Across all the years of record, concentrations of NFR and TP are correlated very strongly in the Elk River ($r = 0.966$, $p < 0.05$) but only weakly in the Kootenay River ($r = 0.110$, $p < 0.05$). This suggests that the distributions of TP in the two rivers are fundamentally different. In the Elk River, where natural sources dominate TP income to the river, most phosphorus is particulate, probably associated with suspended organic matter or soil particles. In the Kootenay River, proportionately much more of the TP would have been dissolved, as a result of enriched effluent discharges from Cominco's fertilizer plant. Since this source was more or less constant, it weakens the natural correlation between levels of suspended solids and TP. Recycling of supernatant from the gypsum ponds, and other improvements in wastewater treatment in the late 1970's, drastically reduced Cominco's TP load to the river. Indeed, the correlation of NFR with TP for data up to 1979 is not even significant whereas for data

Table 4.3.1 Time Series Analysis of NFR and Fluoride in Tributaries
NS = Not Statistically significant (alpha = 0.05) using Kendall Seasonal Tau

Site	Variable	Period	Trend	Slope (mg/L/yr)	Median	Difference
Kootenay R.	NFR	1972/7–1989/10	NS			
	F	1975/6–1989/5	decrease	–0.0074	1974/1–1975/5 1976/1–1977/5	-3.143
Elk R.	NFR	1972/7–1989/11	decrease	–0.14	1977/1–1982/12 1983/1–1987/12	-2.75
	F	1972/1–1989/5	decrease	–0.0024	N/A	

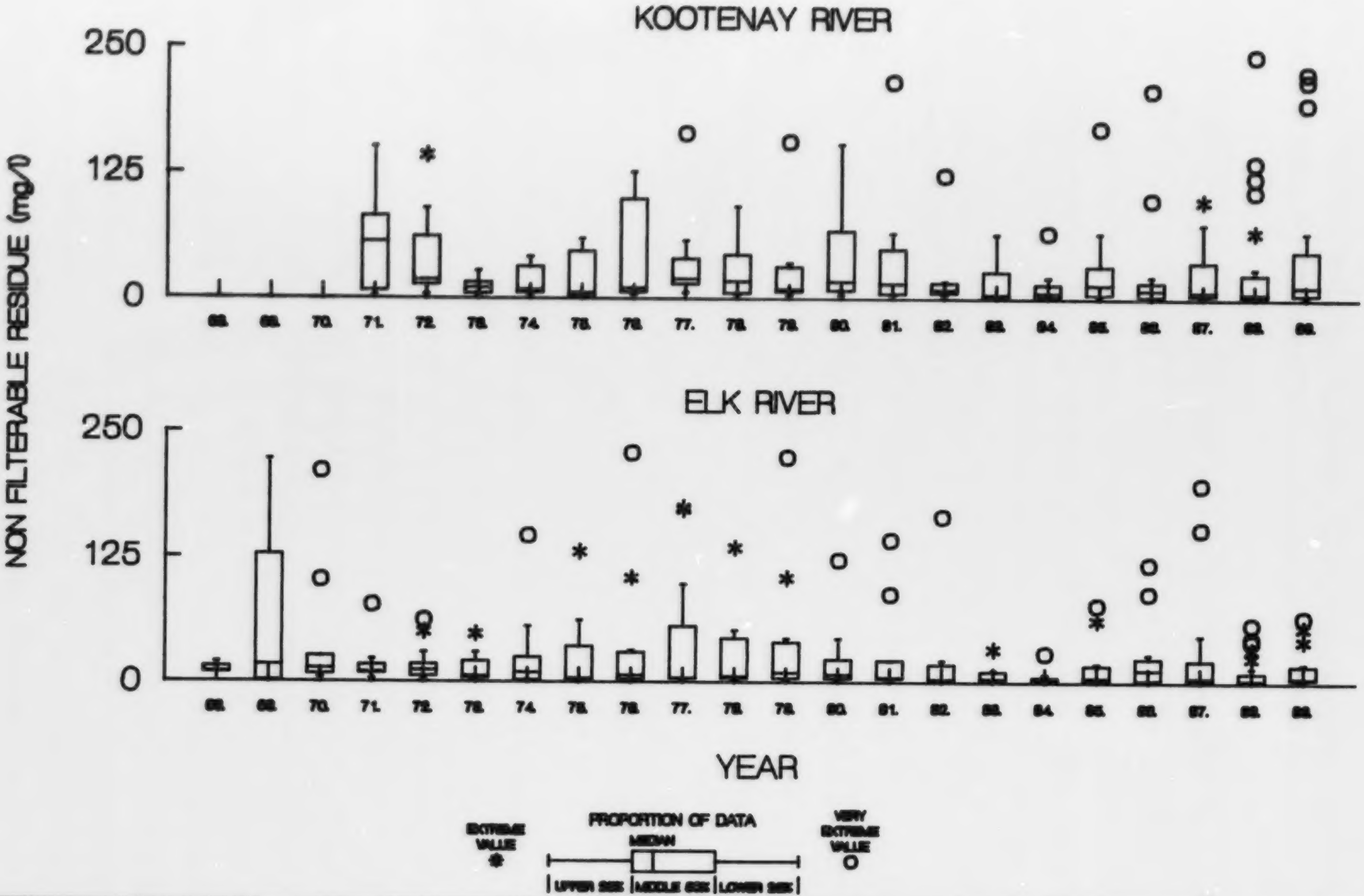

Figure 4.3.2	ANNUAL INFLOW CONCENTRATION NON FILTERABLE RESIDUE	HydroQual Canada Ltd.

after 1979 the correlation is strong and positive (r = 0.923, p <0.05), the same as in the Elk River. Hence, elimination of TP contributions from Cominco Kimberley Operations appears to have restored the natural phosphorus regime in the Kootenay River.

There have been significant reductions in fluoride concentrations in both the Elk and particularly the Kootenay Rivers since the early 1970's (Table 4.3.1). Elk River concentrations are very low compared with those in the Kootenay River (note difference in scale of Figure 4.3.3) and declined very slowly and more or less steadily throughout the 18 years of monitoring (Figure 4.3.3). In contrast, concentrations in the Kootenay River were a full order of magnitude greater than in the Elk River in the early 1970's (Figure 4.3.3), and dropped abruptly after 1975: the difference in median concentrations before and after 1975 was 3.14 mg/L (Table 4.3.1). This was followed by a long, slow decline at a rate similar to that in the Elk River (Table 4.3.1). The decrease in the Kootenay River reflects improved treatment of wastes at the Cominco fertilizer complex. Prior to 1975, the quantity of fluoride discharged to the river was in the order of 340 kg/d (Rocchini, 1981). Improvements in wastewater treatment in 1975 (and again in 1978) greatly reduced this loading. Closure of the plant in 1987 completely eliminated fluoride waste discharge, but fluoride concentrations were already very low by this time.

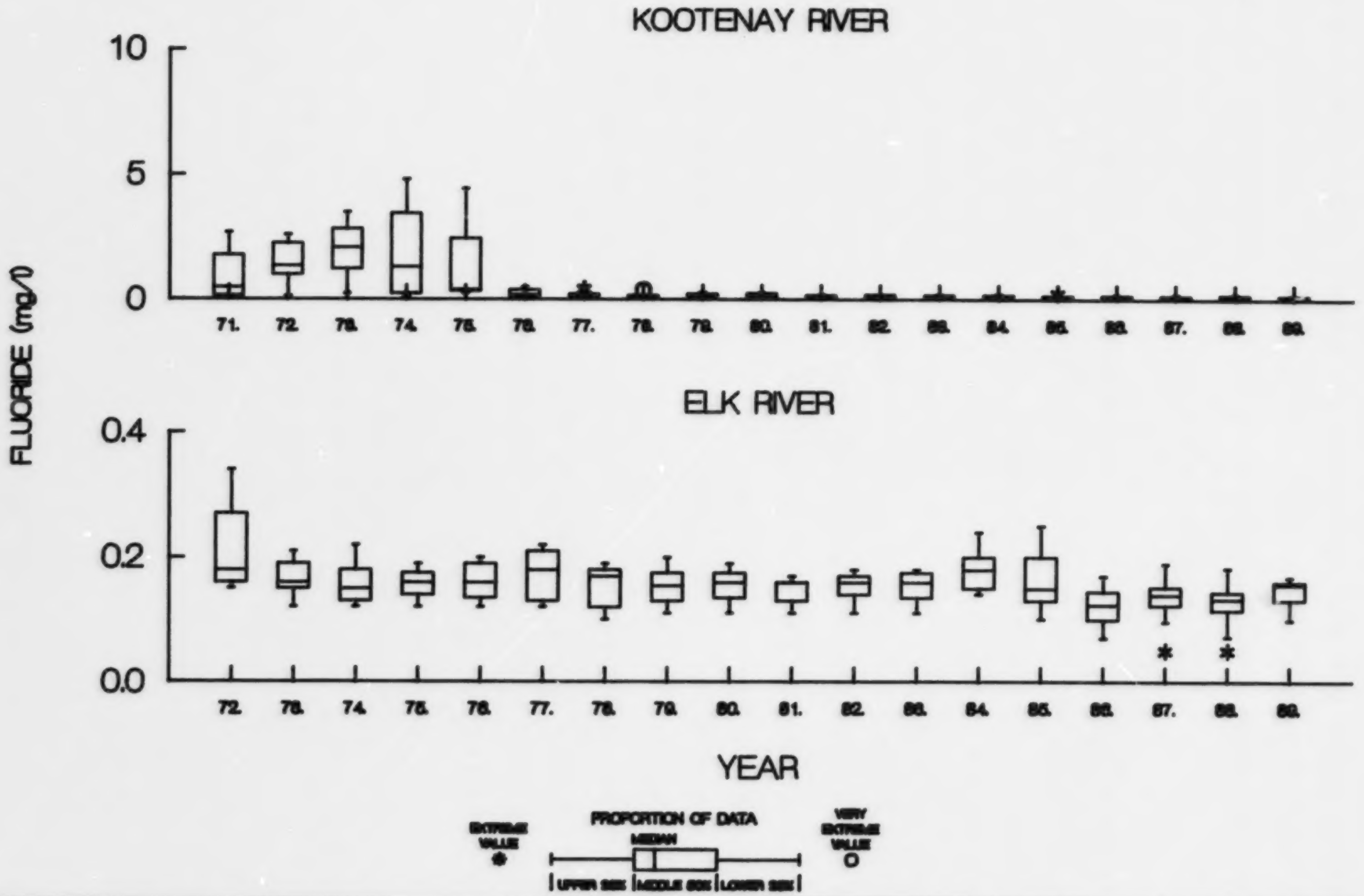

Figure 4.3.3

ANNUAL INFLOW CONCENTRATION
FLUORIDE

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5.0 CHEMICAL ENVIRONMENT IN THE RESERVOIR

5.1 Metals

Monitoring data for selected heavy metals at the International Border site on Lake Koocanusa are summarized in Table 5.1.1. Analysis of these data is confounded by changing analytical sensitivity and a high proportion of detection-limit values. Less than half the samples had detectable copper, and proportions were even less for lead (28%) and zinc (14%). Arsenic and cadmium were never detected in >75 samples (Table 5.1.1). When readings are consistently this low, meaningful averages are difficult to calculate, and any assessment of long-term trends becomes problematic.

It is possible however, to compare reservoir concentrations with those in the tributaries, the Elk and Kootenay Rivers. Of the four metals for which there are sufficient data to derive a meaningful average, all are much lower in concentration in the reservoir than in the tributaries (Table 5.1.2). Reservoir concentrations of Fe and Mn tend to be roughly a third of concentrations in the rivers, and the proportions for Cu and Pb are even less (Table 5.1.2). This reduction arises from sediment-bound metals, usually a substantial part of the total metal concentration, settling out in the still water of the reservoir. Although we have attempted no strict comparison for zinc, the large number of less than detection values for Zn in the reservoir confirms that this metal settles as well. Fifteen of the 19 real measures of Zn in the reservoir were obtained near the sediment. It is evident, therefore, that the reservoir forms an excellent sediment trap for inflowing rivers.

Table 5.1.1 Summary of Detectable Total–Metal Readings in Lake Koocanusa, Border Site. Where multiple analytic methods used concurrently, results for only most sensitive method are reported. Means and standard deviations calculated using non–detectable concentrations set at the detection limit.

Metal	Time Period	Detection Limit mg/L	# of Observations	# Above Detection Limit	Maximum Readings mg/L	Mean mg/l	SD
As	1972/7-1981/6	0.005	24	0			
	1982/7-1986/10	0.25	52	0			
Cd	1982/7-1989/9	0.01	78	0			
Cu	1977/7-1989/9	0.001	128	48	0.008	0.0015	0.0011
Fe	1973/8-1978/10	0.01	26	26	0.37	0.113	0.082
	1974/5-1981/10	0.1	96	51	4.3	0.29	0.59
	1982/7-1989/9	0.01	77	67	0.65	0.155	0.164
Mn	1982/7-1989/9	0.001	63	55	0.035	0.0078	0.0073
Pb	1977/7-1989/9	0.001	129	36	0.011	0.0017	0.0015
Zn	1974/8-1989/9	0.005	133	19	0.024	0.0057	0.0027

Table 5.1.2 Comparison of Total–Metal Mean Concentrations in Lake Koocanusa (Border Site) with Those in the Two Major Tributaries. Numbers in brackets are the Standard Deviations.

		Mean Concentrations (mg/L)		
Metal	Time Period	Elk R.	Kootenay R.	Reservoir
Fe	1973/8–1973/10		0.33 (0.09)	0.11 (0.08)
	1974/5–1981/10	0.81 (1.52)	1.10 (1.70)	0.29 (0.59)
	1982/7–1989/9	0.30 (0.75)	0.60 (1.12)	0.16 (0.16)
Mn	1982/7–1989/9	0.010 (0.03)	0.025 (0.038)	0.0078 (0.0073)
Cu	1977/7–1989/9	0.004 (0.007)	0.005 (0.006)	0.0015 (0.0011)
Pb	1977/7–1989/9	0.012 (0.038)	0.027 (0.034)	0.0017 (0.0015)

5.2 Nutrients

Phosphorus

Reductions of TP concentrations at the Border site (Figure 5.2.1) paralleled those observed in the Kootenay River (Figure 4.2.1). After improved waste treatment began at Cominco's fertilizer plant in 1975, there was a tremendous reduction in TP concentrations at the International Border site; the median concentration for the four-year period before 1975 was 0.026 mg/L higher than the four-year period after 1976. After this major reduction, there was very little year-to-year variation in TP concentrations, and no significant long-term trends could be detected (Table 5.2.1).

No statistically significant long-term trends could be detected in the Forebay data since 1978 (Table 5.2.1); however there was a noticeable reduction in TP concentrations after 1977 (Figure 5.2.1). This was consistent with a significant decreasing trend in SRP concentrations from 1973 to 1989. Compared with data from the Border, the start of this reduction at the Forebay site did not correspond as well to the start of waste-discharge recycling by Cominco Sullivan Mines in 1975: maximum TP concentrations at the Forebay site are lower than at the Border site, and decline later (Figure 5.2.1). This is to be expected for an inflow whose effect would be more diffuse toward the dam as it mixes with an increasing volume of reservoir water and undergoes sedimentation. This implies that water in the downstream end of the pool must have a residence time exceeding a year, considerably greater than the mean residence time.

Although average TP concentrations over the entire study period were significantly higher at the Border site than at the Forebay site, the difference is very small (0.002 mg/L) and probably reflects progressive loss of phosphorus through sedimentation.

A monthly breakdown illustrates how TP and SRP concentrations were distributed vertically in the water column during the open-water season at both the Border and

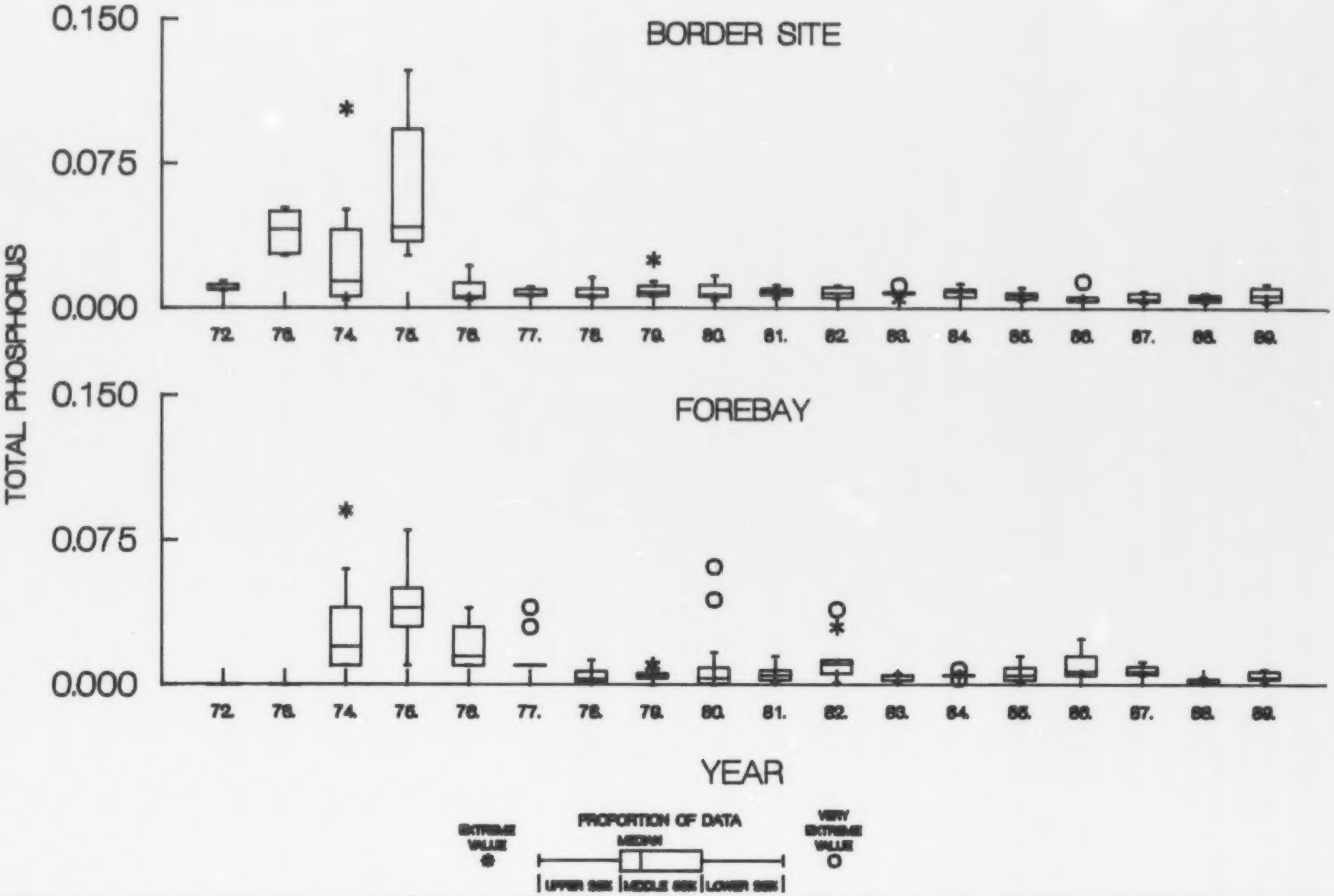

Figure 5.2.1	ANNUAL EUPHOTIC ZONE (0–10m) TOTAL PHOSPHORUS CONCENTRATIONS	HydroQual Canada Ltd

Table 5.2.1 Long Term Trends for Lake Koocanusa Border Site and Forebay, as indicated by Seasonal Kendall Tau (alpha = 0.05); NS = Not Statistically Significant; Slope refers to the rate of change (mg/L/yr); Median refers to a Wilcoxin signed rank test comparing median between two adjacent time periods; Date refers to the time periods that were compared; and Difference indicates the difference between the median of the two periods. N/A = not applicable. Data for top 10 m only.

		Seasonal Kendall Tau			Median	
Site	Constituent	Period	Trend	Slope	Date	Difference
Border	TP	1978–1989	NS			
	OP	1978–1989	NS			
	TN	1972–1989	decrease	-0.0033	N/A	
	TKN	1974–1989	decrease	0.002	N/A	
	NH3	1974–1989	decrease	-0.0002	N/A	
	NO2+NO3	1976–1989	NS			
Forebay	TP	1978–1989	NS		(1974/5–1977/11) (1978/5–1981/11)	-0.019
	OP	1973–1989	decrease	-0.0002		
	TN	1972–1989	increase	0.008		
	TKN	1972–1989	increase	0.007		
	NH3	1973–1989	NS		(1979/05–1980/08) (1981/05–1982/08)	0.183
	NO2+NO3	1973–1989	N/A		N/A	

Forebay sites (Figures 5.2.2 and 5.2.3). At the Border site, concentrations of both TP and SRP before 1978 were evenly distributed throughout the water column except in August (Table 5.2.2); by comparison, concentrations after 1977 varied significantly with depth from June to September but were uniformly distributed in May.

At the Forebay site, concentrations before 1977 of TP and SRP varied significantly with depth, except in May; however, after 1977, there was generally a uniform vertical distribution of TP and SRP throughout the open-water season except in June and July (Table 5.2.2). Generally, the more recent trend is towards a more even distribution of phosphorus with depth at the Forebay site, and for an accumulation of phosphorus near the sediment at the Border site. The tendency for phosphorus to accumulate at the bottom is consistent with findings of Iskandar and Shukla (1975) that Lake Koocanusa acts as a phosphorus sink. Sedimentation of diatoms, loricated chrysophytes, and dinoflagellates from the euphotic zone, as well as inorganic phosphorus adsorbed to suspended sediments during periods of high tributary flow may account for this accumulation. The even distribution of phosphorus with depth at the Forebay site suggests that water may be released from any depth in Lake Koocanusa without altering the phosphorus export to Kootenay Lake.

Nitrogen

There were significant long-term decreases in TN, TKN and NH_3 concentrations at the Border site (Table 5.2.1), which corresponds with the reduced loading from the Kootenay River. By comparison, there were significant increases in TN and TKN further downstream at the Forebay site (Table 5.2.1). The rate of increase is not large, about 2% per year. The rate of increase for TKN is nearly identical to the rate for total nitrogen; that, along with the lack of a long-term trend for ammonia suggests that increases in organic nitrogen are driving the total nitrogen increase.

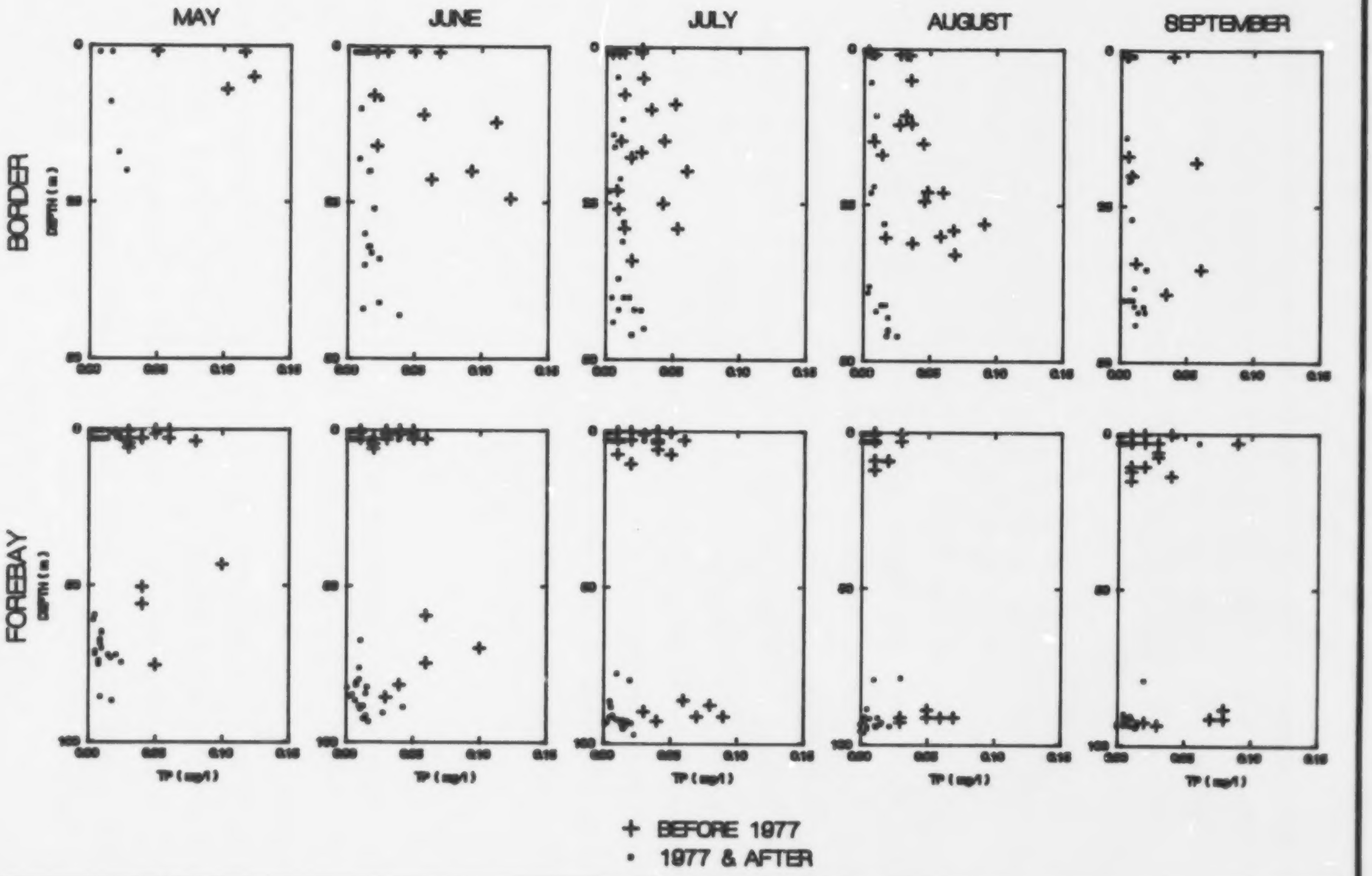

Figure 5.2.2

TOTAL PHOSPHORUS DEPTH PROFILES
BORDER AND FOREBAY

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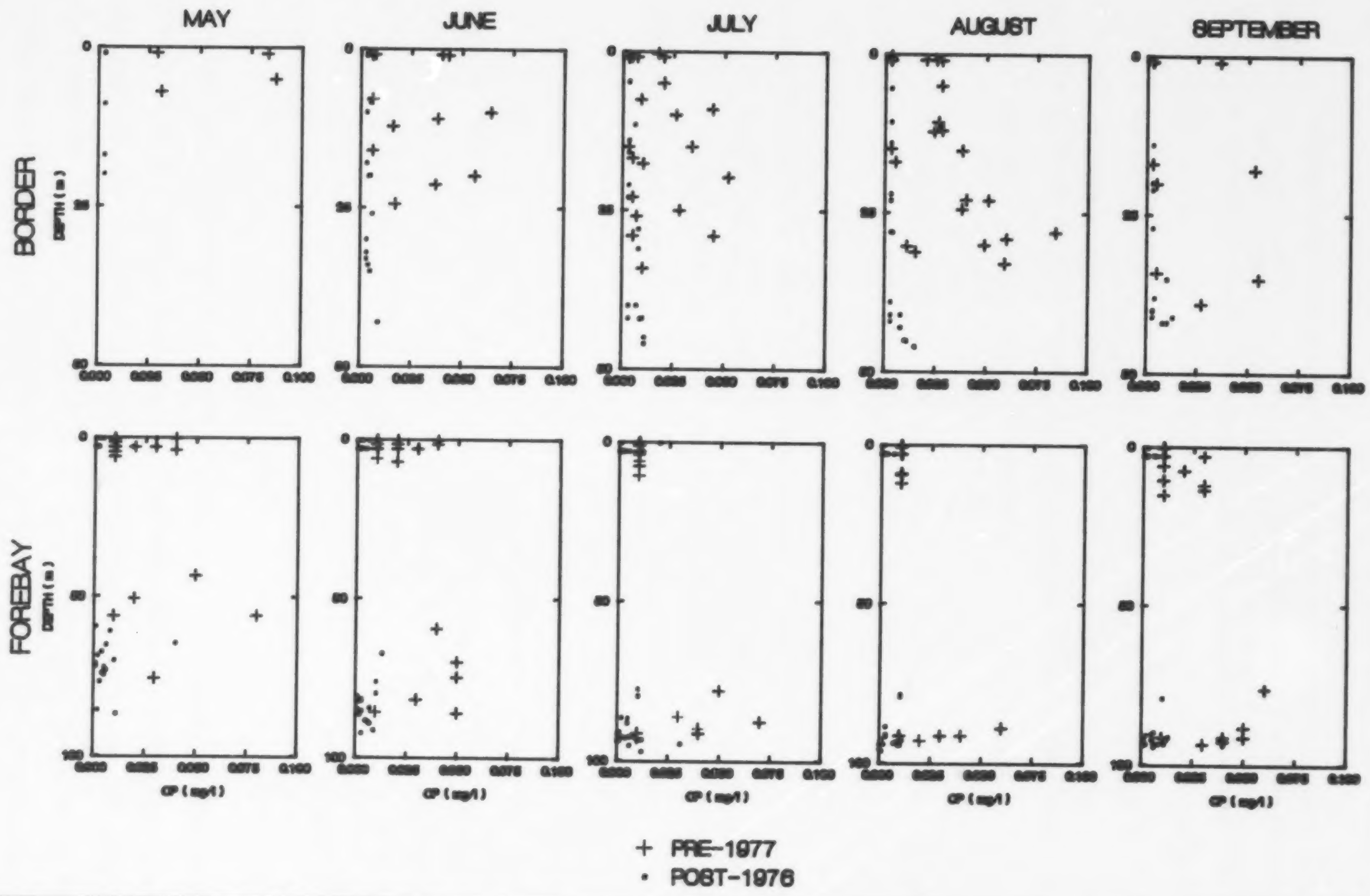

Figure 5.2.3

SOLUBLE REACTIVE PHOSPHORUS DEPTH PROFILES
BORDER AND FOREBAY

HydroQual
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Table 5.2.2 Test for Significant Differences in Total Phosphorus and Orthophosphorus with Depth in Lake Koocanusa.
Tested using regression of phosphorus concentration on depth. NS – no significant change with depth, SIG = significant change with depth.

		PHOSPHORUS			
		Pre–1978		Post–1977	
Site	Month	Total	Ortho	Total	Ortho
Border	May	NS	NS	NS	NS
	June	NS	NS	SIG	SIG
	July	NS	NS	SIG	SIG
	August	SIG	SIG	SIG	SIG
	September	NS	NS	SIG	SIG
Forebay	May	NS	SIG	NS	NS
	June	SIG	SIG	SIG	NS
	July	SIG	SIG	SIG	NS
	August	SIG	SIG	NS	NS
	September	SIG	SIG	NS	NS

There is no clear source of this added nitrogen. Fixation of atmospheric nitrogen by blue-green algae is the most obvious possibility, but populations of blue-greens are low, and nitrogen-fixation would not be expected to be active in this oligotrophic reservoir. Alternatively, there may be nitrogen mineralization in the sediments and rapid incorporation into phytoplankton. Phytoplankton biomass at this site has been episodically increasing since 1977 (see Section 6.1). It is also possible that the trend is an artifact of changing analytical methods.

In addition to these long-term trends, there was an abrupt increase in ammonia concentrations in 1981 and 1982 which has no apparent antecedent, and which cannot be attributed to a change in analytical technique (Figure 5.2.4); (although the data may simply be erroneous). Since nitrate-nitrite concentrations did not vary significantly from year to year at either sampling station (Table 5.2.1), we have no evidence that increased concentrations of nitrate-nitrite from the Elk River have had any significant effect on ambient reservoir concentrations.

Total inorganic nitrogen (TIN) and TN concentrations tend to decline through the open-water season at the Border site, while concentrations at the Forebay site remain relatively high and constant throughout the season (Figure 5.2.5). The late-season reduction at the Border site may simply reflect the effect of drawdown, which commences in October or November.

Inter-Agency Data Comparison

We compared data collected by U.S.G.S. and B.C. Environment at the Border site between 1978 and 1989 to evaluate the compatibility of the two databases. There were no significant differences between agencies with respect to TP, SRP and nitrate-nitrite concentrations; however, concentrations of TN, TKN and NH_3 measured by the U.S. agency were statistically higher than those measured by B.C. Environment (Table 5.2.3).

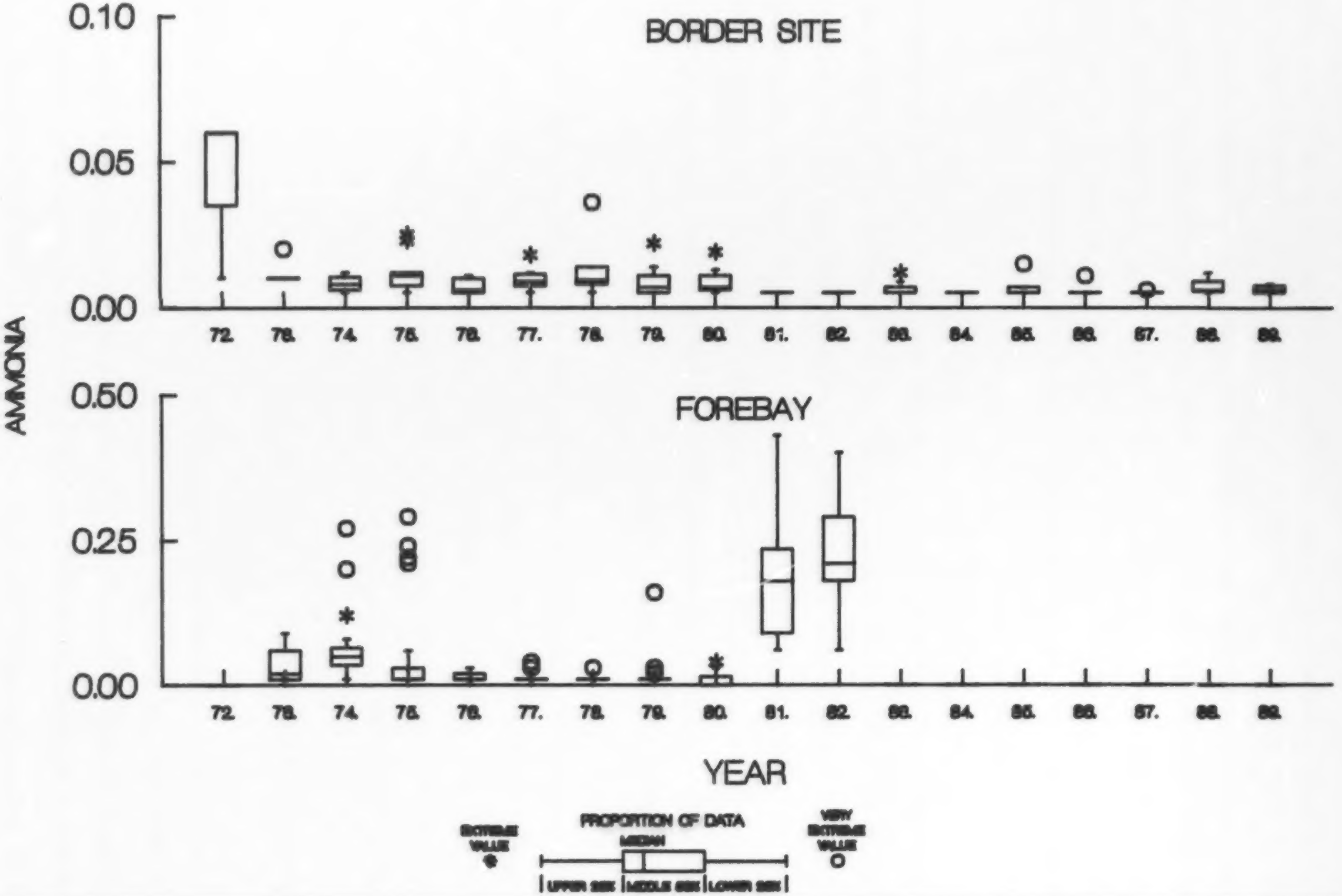

Figure 5.2.4

ANNUAL EUPHOTIC ZONE (0-10m)
AMMONIA CONCENTRATIONS

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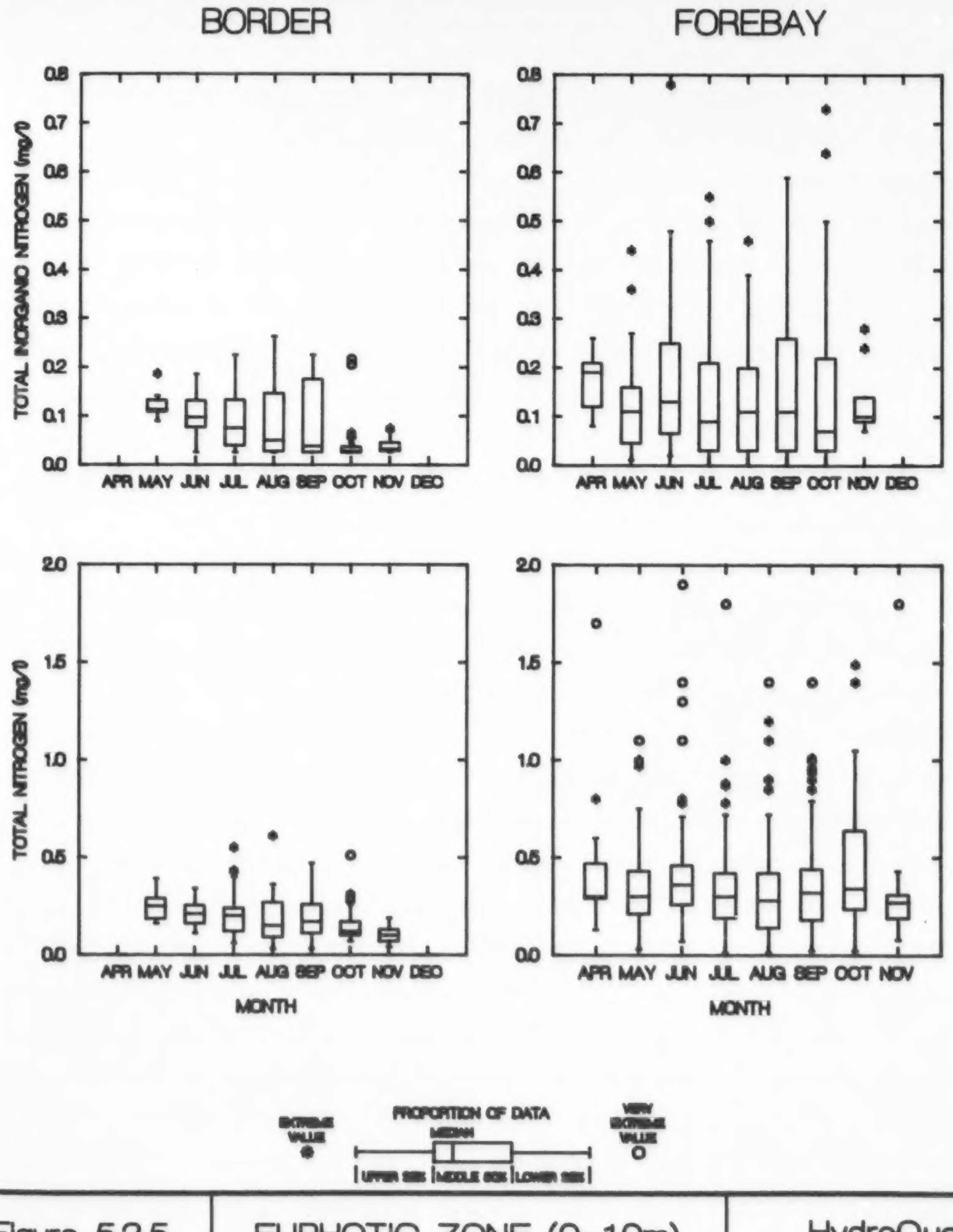

Figure 5.2.5	EUPHOTIC ZONE (0–10m) NITROGEN	HydroQual Canada Ltd.

Table 5.2.3 Comparison of Medians of U.S.G.S. and B.C. Environment Nutrient Data at Border Site. NS = Not Significant, SIG = Significant. Difference indicates how much lower the Canadian values are relative to American.

Nutrient	Test Result	Difference (mg/L)
TP	NS	-
OP	NS	-
TN	SIG	-0.220
TKN	SIG	-0.157
NO2+NO3	NS	-
NH3	SIG	-0.007

The principal source of error lies in analysis of the organic portion of total Kjeldahl nitrogen; the discrepancy with respect to ammonia is too small to matter, and NH_3 composes only a small part of TKN. The bias in organic nitrogen determinations, in turn, appears to be largely responsible for the discrepancy in measurements of TN.

The magnitude of the difference for TN and TKN approaches 100% of typical concentrations in the reservoir. Furthermore, it is impossible to say, from the data available, which agency's measurements are more accurate, or to derive a correction factor to conflate the two databases. This problem needs to be resolved, through a careful comparison of blind analyses of standard samples by both agencies, before the full value of monitoring data can be realized.

TN:TP Ratios

The ratio of TN to TP (TN:TP) can be used as an index of nitrogen versus phosphorus limitation for algal growth. Generally values in excess of 12 (dotted line in Figure 5.2.6) indicate phosphorus is limiting, whereas very low values, i.e. less than 7, infer nitrogen limitation. Owing to the generally higher nitrogen concentrations at the Forebay site, the corresponding TN:TP ratios were also higher (with medians near 50), and suggest that this part of the reservoir is strongly phosphorus limited (Figure 5.2.6). Because U.S.G.S. measurements of TN are systematically greater than B.C. Environment measurements, the median TN:TP ratios could be as low as 20 if the U.S. data are in error. However, this would still infer phosphorus limitation in the lower reservoir. Evidence for phosphorus limitation is less conclusive at the Border site, where based upon the Canadian data, median TN:TP ratios exceed 12 only in July through October, and seldom exceed 20 (Figure 5.2.6). Again however, if B.C. Environment measurements of TN are biased downward (rather than U.S.G.S. data being biased upward) then the true TN:TP ratios at this site would be much greater, and clearly indicative of phosphorus limitation. Until the issue of data accuracy is resolved, discussions of N or P limitation in the reservoir must remain speculative.

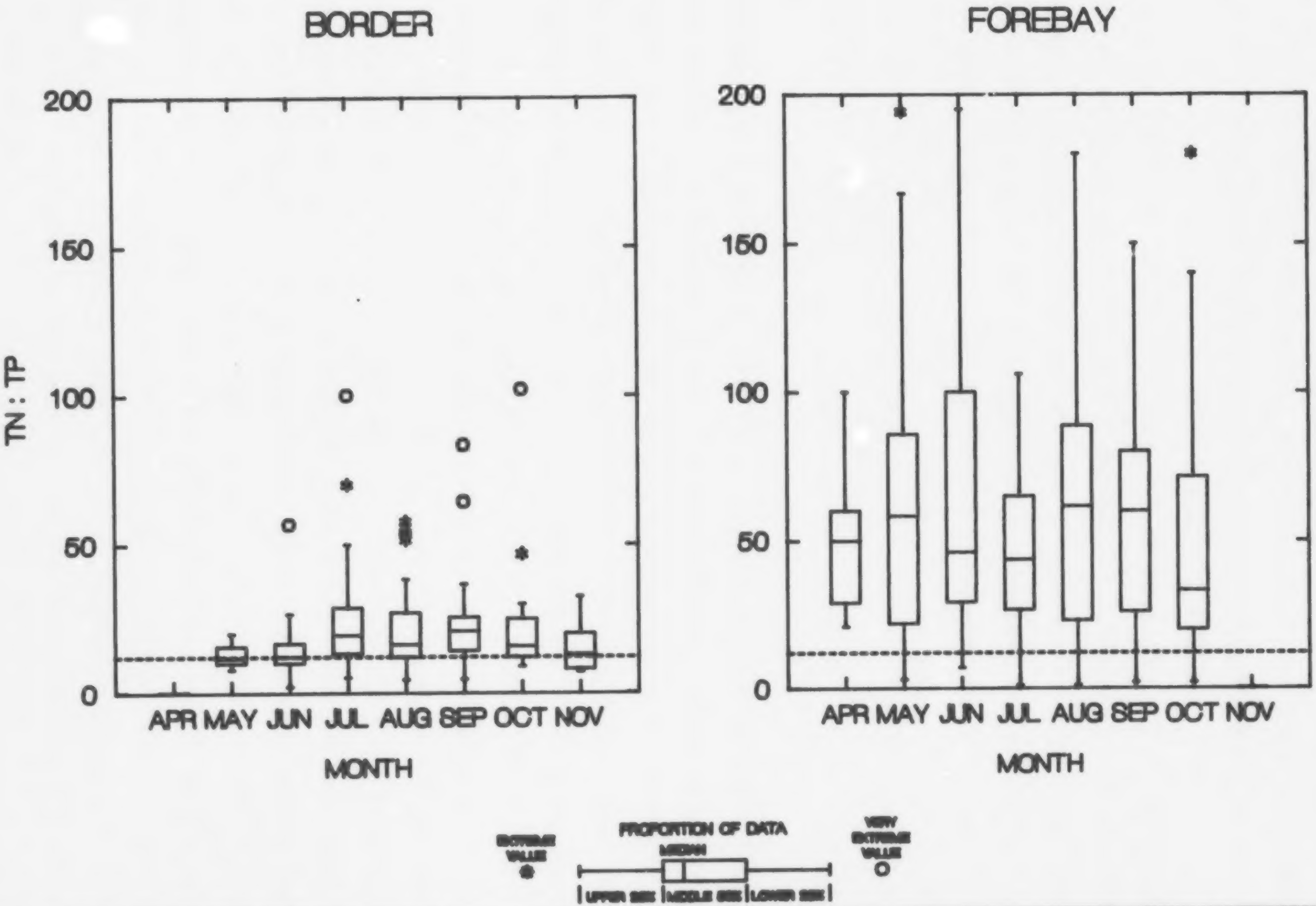

Figure 5.2.6

TN : TP RATIOS
LAKE KOOCANUSA

HydroQual
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5.3 Other Parameters

Although pH in the reservoir has increased significantly from 1972 to 1989 (Table 5.3.1; Figure 5.3.1), the rate of increase is only slightly more than 0.01 pH unit/year for a total increase of 0.2 pH units over the 18 years. This marginal increase may simply reflect normal changes in the reservoir's water chemistry as it equilibrates with the underlying sediments. It may as well reflect pH increases in the tributaries, particularly the St. Mary River, from the elimination of acid mine drainage in the 1970's. The range of pH values observed is well within that for water bodies dominated by calcium bicarbonate. During the open-water season, pH did not vary significantly from month to month and tended to remain between 8.0 and 8.5 pH units in an average year (Figure 5.3.2). Some of the variation in pH undoubtedly results from mixing of field and laboratory measurements in the database.

Since there were no TDS data from June 1977 to 1981, it was inappropriate to run a trend analysis through the remaining years; therefore, a median test was performed to compare the period from 1972 to 1977 with 1981 to 1989. This test indicated that TDS concentrations have declined slightly, by 9.1 mg/L, a difference of marginal biological significance. No significant trends were noted in total alkalinity between 1972 and 1981 (Table 5.3.1). During the open-water season, alkalinity tended to decline slightly in July, which probably corresponds to increased primary productivity and the accompanying uptake of carbon from the water column (Figure 5.3.2). Otherwise, alkalinity was very stable over the ice-free season. Specific conductance also appears to increase slightly as the season progresses (Figure 5.3.2). This probably reflects the slightly decreasing influence of dilute tributary flow from spring through summer and fall, as flows diminish and the reservoir fills.

There were insufficient sulphate data (1972 to 1977) to conduct a trend analysis. In an

Table 5.3.1 Long–Term Trends for Routine Constituents at Lake Koocanusa Border Site. Trend tested using Seasonal Kendall Tau or Wilcoxin signed rank test (alpha = 0.05). NS = No Significant trend, N/A = Not Applicable; Slope = rate of change in mg/L/yr; Dates indicate time periods compared with Wilcoxin text, Difference = the difference between the medians of the two periods compared. Concentrations tested were means over all depths.

	Seasonal Kendall Tau			Median Test	
Constituent	Period	Trend	Slope	Dates	Difference
T.D.S.		NA		1972/11–1977/5 1981/11–1989/5	9.1
pH	1972/7–1989/5	increase	0.012		
Total Alkalinity	1972/8–1981/8	NS			

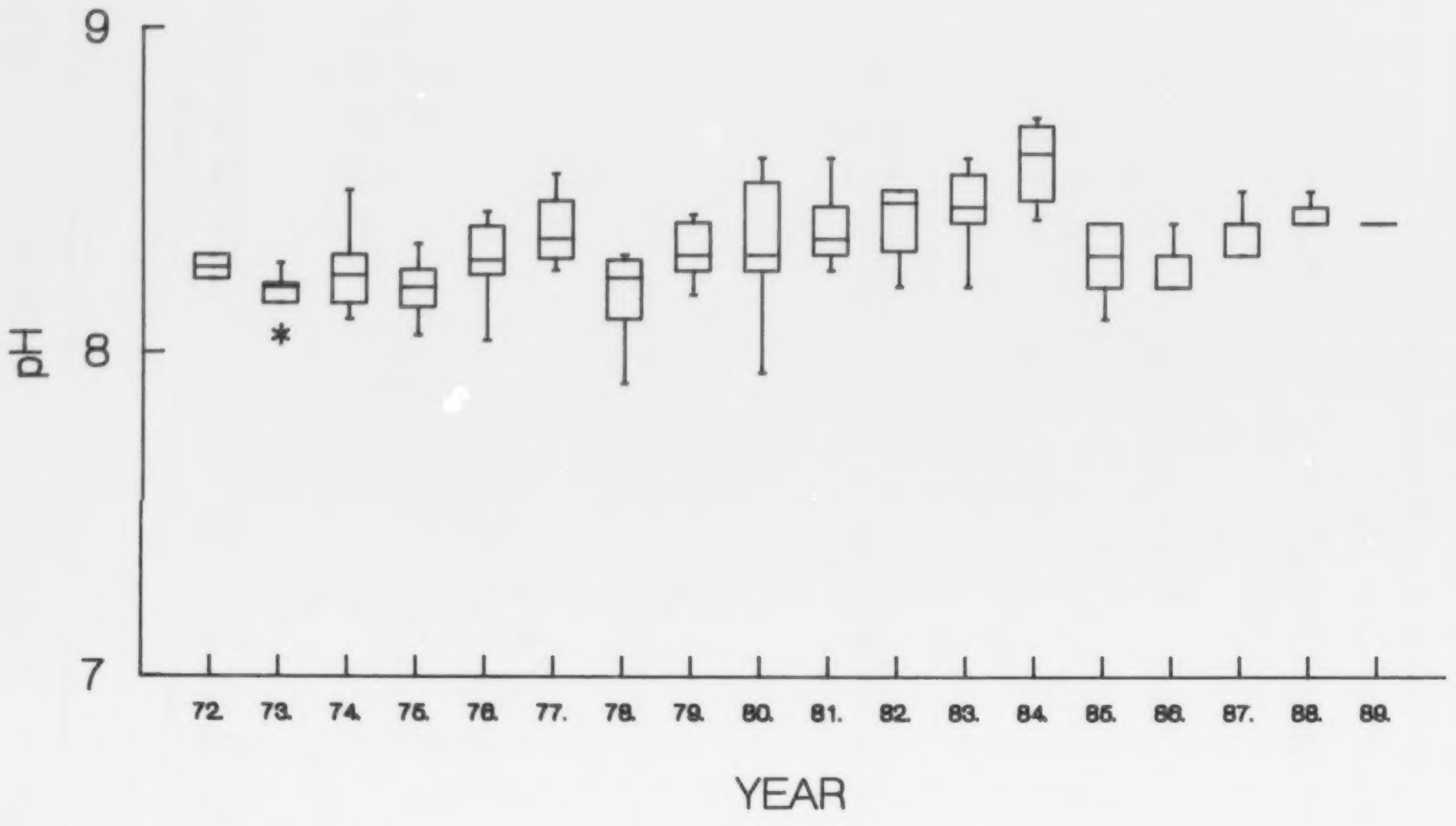

Figure 5.3.1

ANNUAL TREND IN RESERVOIR pH

BORDER SITE

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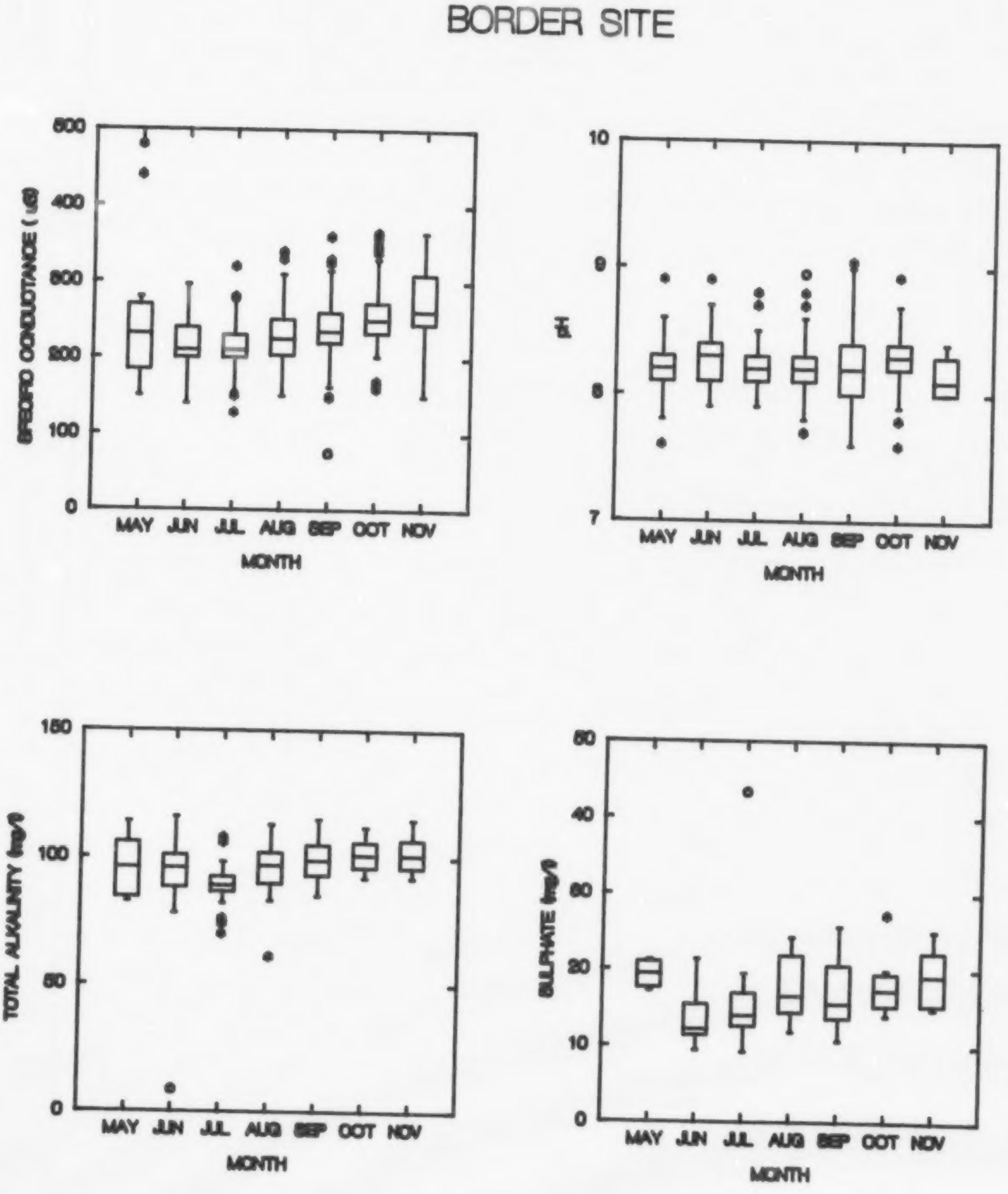

Figure 5.3.2	SEASONAL PATTERN IN SELECTED CONSTITUENTS	HydroQual Canada Ltd.

average year, May concentrations are high, probably as a result of sulphate carried over from the previous season; this is followed by a relatively rapid decline to its seasonal minimum in June, as tributary water dilutes the reservoir sulphate. Concentrations increase slightly through the season probably through evaporation and biological retention, until in November concentrations approach those observed in May (Figure 5.3.2).

6.0 BIOLOGICAL ENVIRONMENT IN THE RESERVOIR

6.1 Phytoplankton

6.1.1 Long-term Trends

Chlorophyll a measured at the Border site varied without any significant trend from 1973 to 1988 (Figure 6.1.1). Annual means generally ranged between 1.0 and 2.0 μg/L, except in 1976 and 1984, when they approached 3.0 μg/L. By comparison, the concentration of phytoplankton biomass remained <100 μg/L during the first four years of impoundment; after 1977, concentrations increased above 100 μg/L and tended to remain at this level for most of the remaining years, again excepting 1984, when algal biomass exceeded 600 μg/L. The sharply greater algal biomass in 1984 was not caused by higher phosphorus levels, although it was associated with a jump in pH that year (see Section 5). The 1975 and 1976 average chlorophyll values for the Forebay site and the Border site were similar. Over the 1977 to 1982 interval, values were slightly lower at the Forebay site. Chlorophyll data have not been collected at the Forebay site since 1982.

From 1975 to 1988 there were large year-to-year variations in the percent composition of major algal groups in the reservoir (Figure 6.1.2) (1989 data shown are from a single sample). For example, the large algal biomass in 1984 was composed almost exclusively of the pennate diatom Tabellaria fenestrata and the colonial chrysophyte Dinobryon divergens, both of which were common forms in the reservoir in other years. By comparison, the dinoflagellate, Ceratium spp. formed a larger proportion of the sample biomass in 1979, 1980 and 1986 than in other years, and completely dominated samples taken in 1975. This does not necessarily indicate they were more abundant during these years, however because the distribution of these large organisms (equivalent in mass to 70 Tabellaria and 140 Dinobryon) is usually very uneven, and their chance occurrence in some samples can greatly bias results. Centric diatoms,

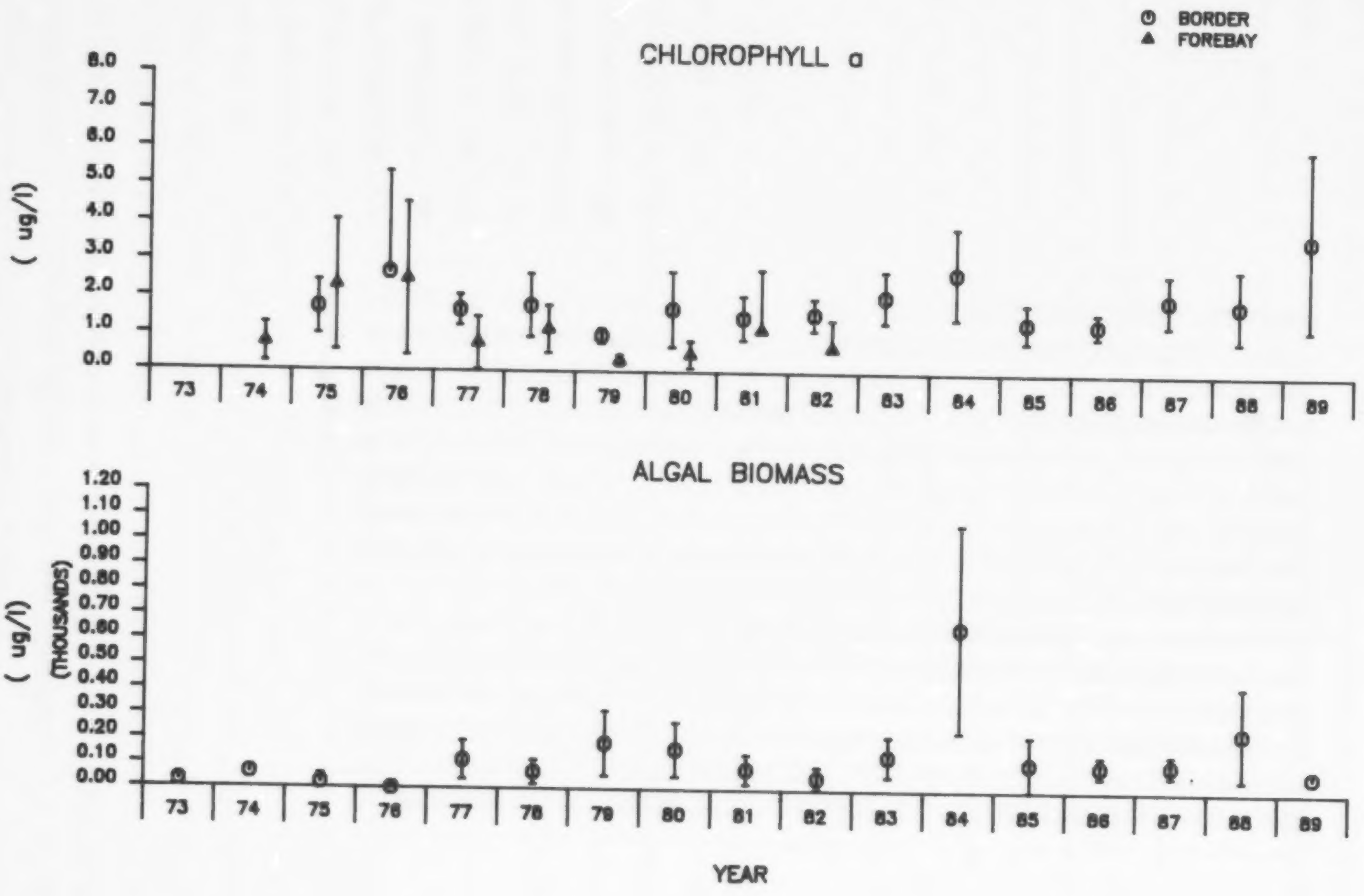

Figure 6.1.1

LONG TERM PATTERNS
IN ALGAL ABUNDANCE

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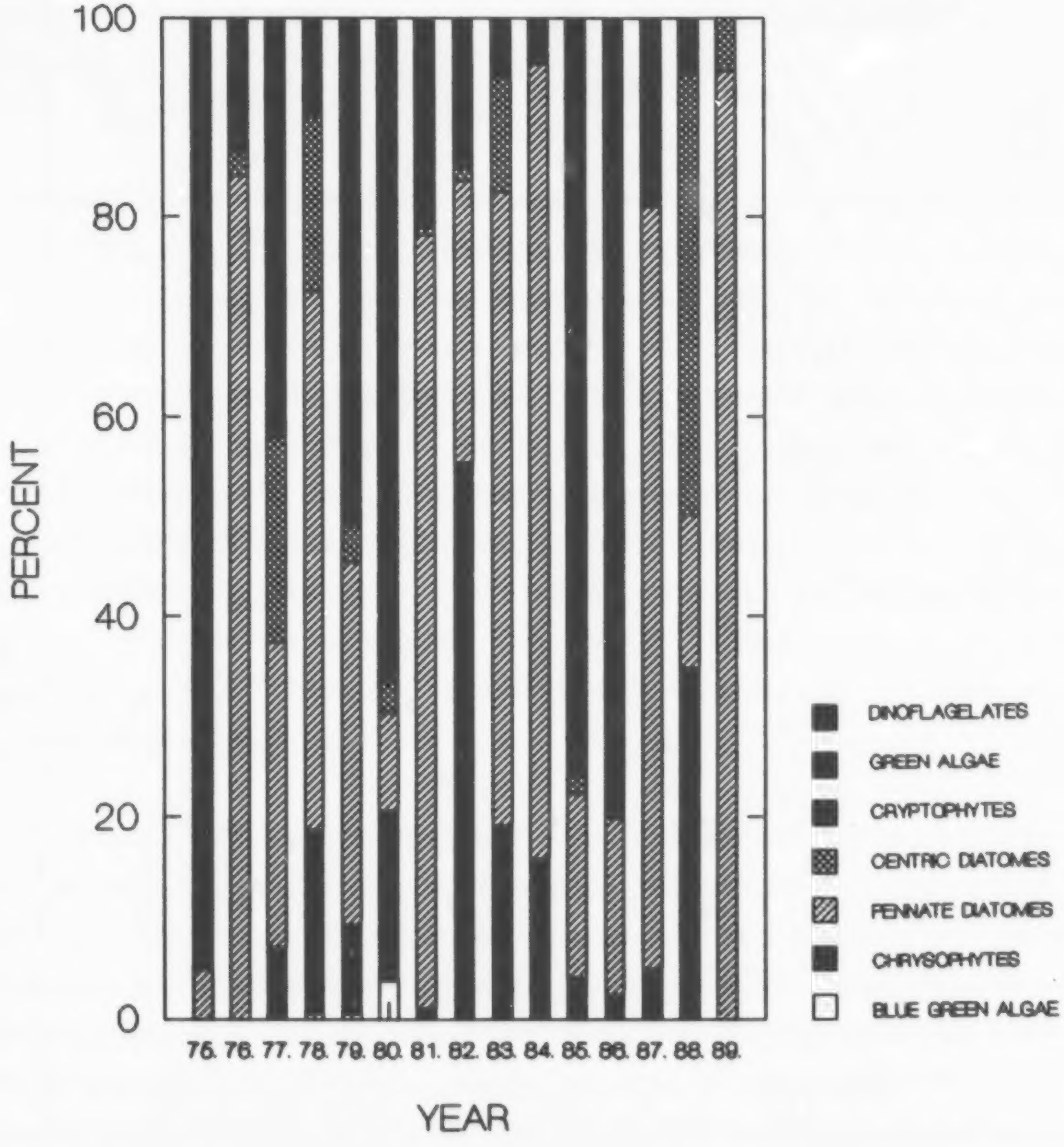

Figure 6.1.2

AVERAGE PHYTOPLANKTON BIOMASS
(JUNE - AUGUST)

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notably Melosira sp. and Cyclotella sp., also alternated in abundance with pennate diatoms, and were prevalent in 1977, 1978, 1981 and in the more recent years. The least abundant forms were the green and blue-green algae, neither of which constituted more than 10% of the community biomass in any year.

Crozier and Nordin (unpublished) noted that the dominant pennate diatoms in the early years of impoundment (1973-1975) were Asterionella spp. and Fragilaria spp. These genera continued to be common in subsequent years, although their former prominence has been eclipsed by Tabellaria. Crozier and Nordin also noted the dominance of ensheathed algal genera such as Gleocystis, Chroococcus, Aphanothece as well as the filamentous blue-greens, Anabaena and Aphanizomenon at different times during those early years. Except for these latter filamentous algae, the colonial blue-green algae continued to be common between 1978 and 1988.

There is no consistent change in community structure to account for the discrepancy between chlorophyll a and algal biomass measurements in 1975 and 1976. Large dinoflagellates dominated the biomass in 1975, which may have changed the community chlorophyll-to-biomass ratio, but 1976 samples were dominated by pennate diatoms, which were abundant in most other years.

6.1.2 Seasonal Distribution

The distribution of phytoplankton biomass from June to October shows a maximum in mid-summer, with much lower concentrations in spring and fall (Figure 6.1.3). The corresponding seasonal profile of chlorophyll a concentrations confirms the biomass maximum in July, although that is also the month with the greatest variance in chlorophyll concentrations. These results are consistent with studies of Woods (1979) and Chisholm et al. (1989) in which peak primary productivity was recorded in July and August, when surface waters are warmest and the euphotic zone extends to its greatest depth.

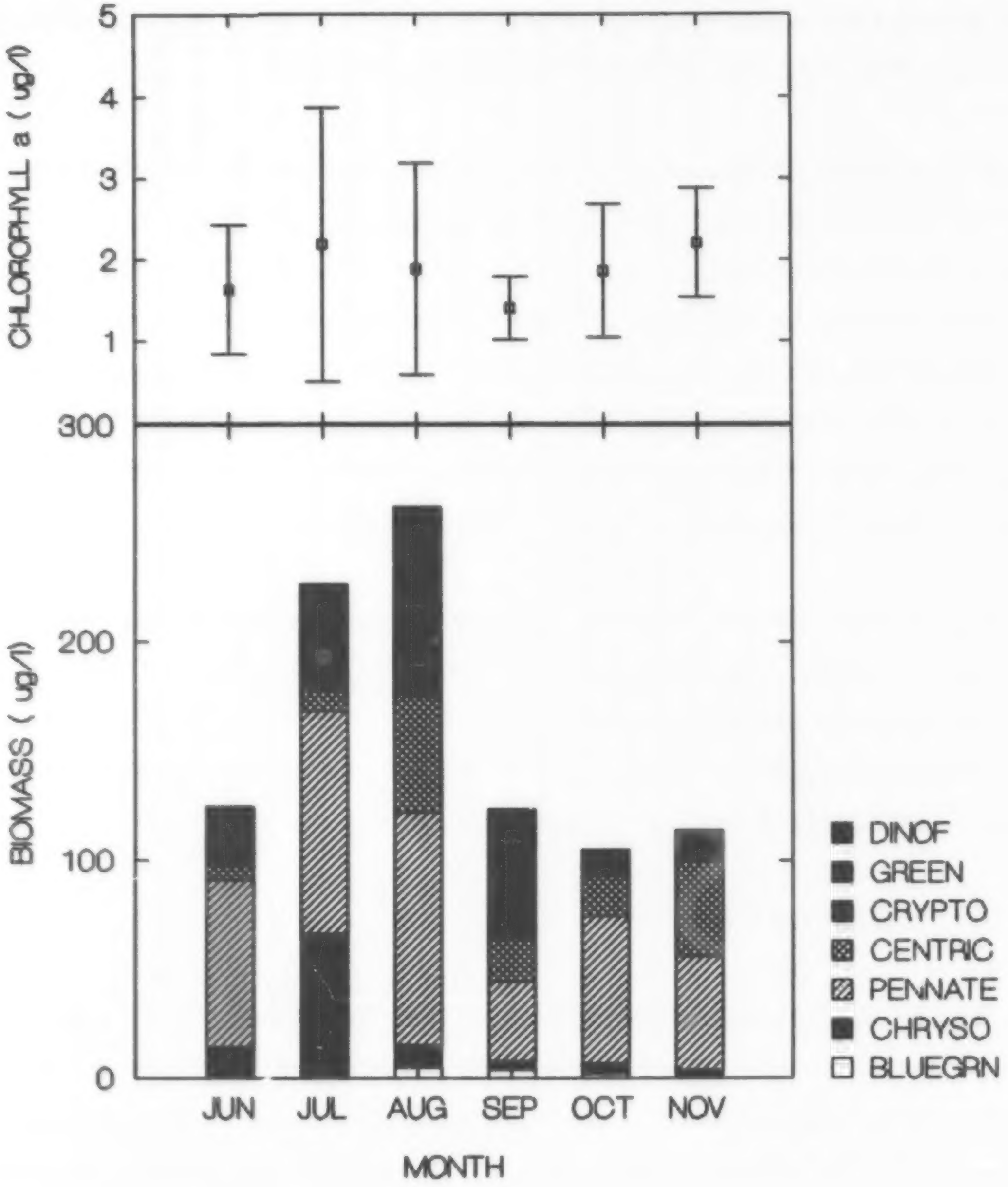

Figure 6.1.3	SEASONAL PATTERNS IN PHYTOPLANKTON	HydroQual Canada Ltd.

Averaging species abundance data over the 1978 to 1988 period indicates that green algae and chrysophytes are most abundant in July but rare in other months. The pennate diatoms are abundant throughout the summer, especially July and August, but form a small proportion of the late spring (June) and fall biomasses.

Woods (1979) concluded that the dominant influence on primary production in Lake Koocanusa was the quantity of light, although this in turn was controlled by at least three factors: the thermal structure of the reservoir, which allows phytoplankton to be circulated out of the euphotic zone; turbidity in the inflows during spring runoff; and seasonal and meteorological variations in incident solar radiation. Woods and Falter (1982) also concluded that seasonal variations in primary production could not be explained by ambient concentrations of nitrogen and phosphorus, indicating that the phytoplankton was not nutrient limited. TN:TP ratios (Section 5.2), however infer the possibility of P limitation in the reservoir.

6.2 Zooplankton

6.2.1 Long-term Trends

Prior to 1982, the mean biomass of zooplankton at the Border site was very low (generally <1 g/m^2; Figure 6.2.1). In 1982, community biomass increased an order of magnitude to over 11 g/m^2. Biomass decreased substantially in the next four years, although it remained high relative to before 1982. After 1986, zooplankton biomass declined further to a mean concentration <1 g/m^2. There is very poor correspondence between the timing of this dramatic increase in zooplankton standing crop and trends in algal biomass, indicating that the sudden proliferation of zooplankton is not likely a direct response to any change in the phytoplankton biomass. The apparent change in zooplankton biomass may be exaggerated, however, by spatial and temporal patchiness in the plankton distribution, which is not handled adequately by fixed-station

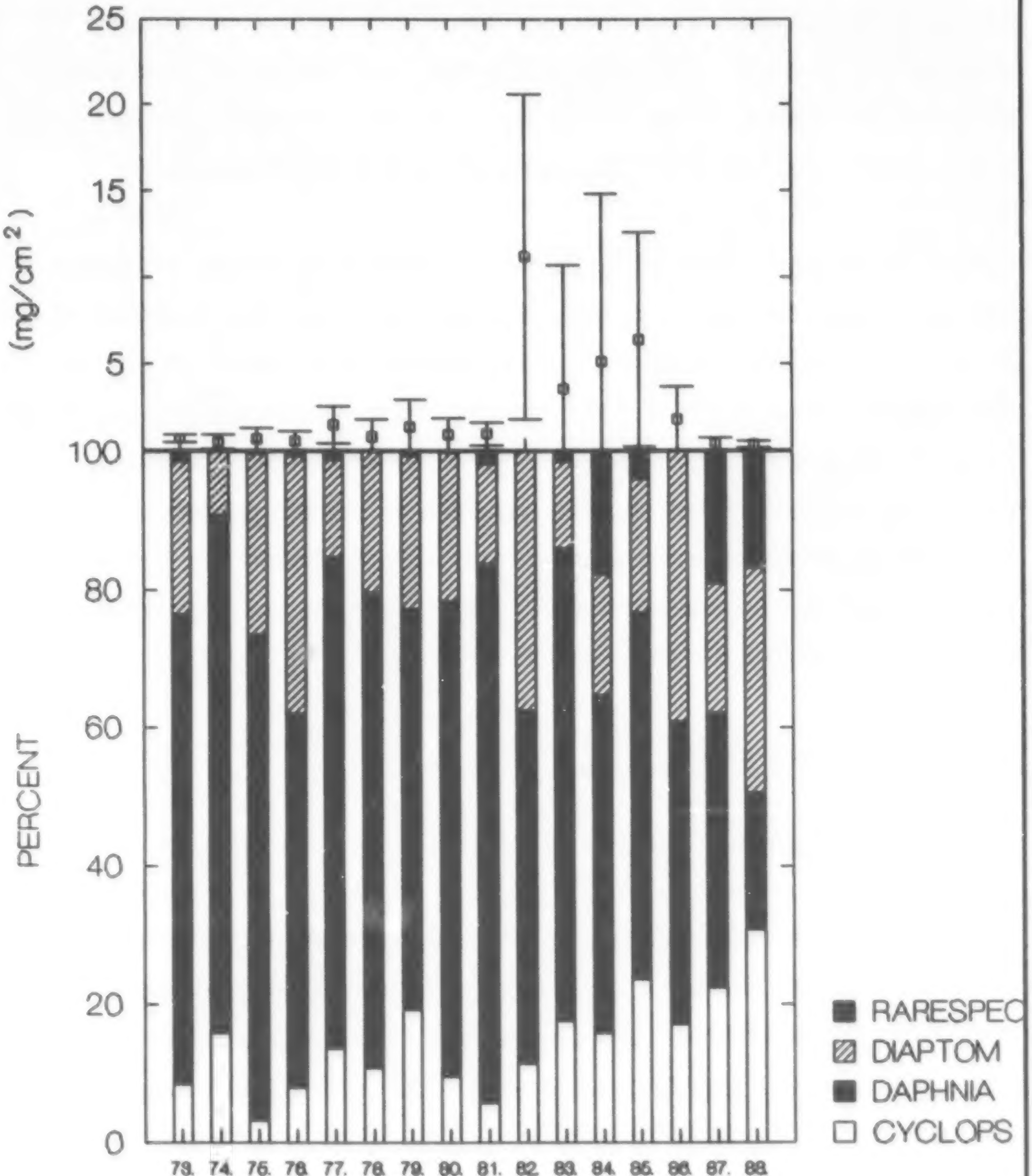

Figure 6.2.1	AVERAGE ZOOPLANKTON BIOMASS AND PERCENT COMPOSITION	HydroQual Canada Ltd

sampling. This problem may be exacerbated by planktivorous fish, such as kokanee, that deplete zooplankton from sections of the lake as they pass through in feeding schools. The abrupt change in zooplankton biomass occurred two years after 250,000 kokanee fry were released into the lake.

In this respect it is significant that, coincident with the 1982 increase in mean zooplankton biomass, there was a proportionately large increase in annual variance, i.e., by a factor of ten. In some months the sampled biomass was very high, in other months relatively low; a few extreme values often skewed the means of these data, which is to be anticipated when sampling an overdispersed population. None of these possibilities, unfortunately, accounts for the abrupt decline of mean zooplankton biomasses to the 1 g/m^2 range again after 1986.

6.2.2 Species Composition

The species composition of the zooplankton have generally remained the same from 1973 to 1988. The three dominant taxa are Daphnia, Diaptomus, and Cyclops; rarer taxa include Bosmina, Epischura, Leptodora, Ceriodaphnia, and Diaphanosoma (Table 6.2.1, Figure 6.2.2).

Of the major taxa, the relative abundance of Cyclops and Diaptomus have increased the most since 1982 (lower panel of Figure 6.2.1). They each accounted for <20% of the total biomass before 1982, but in recent years their proportions have each increased into the 20 to 50% range. Conversely, Daphnia has decreased proportionately: while it composed almost 80% of the total community biomass in the early 1970's, its relative contribution has been substantially eroded (to 50%) since 1982, probably as a consequence of predation by kokanee (see Section 6.3). However, the total Daphnia biomass also increased over the 1982 to 1986 interval (Figure 6.2.2). Hence, the dramatic increase in total zooplankton biomass in 1982-1985 was not the result of a large change in species composition.

Table 6.2.1 Zooplankton Taxa in Lake Koocanusa

Daphnia spp.	In 1978, the National Museum identified two species D. schodleri and D. galeata mendotae. (The MOE usually reported only D. pulex which Ottawa said was very similar to D. schodleri.) Zenon now reports all Daphnia as D. galeata mendotae.
Cyclops bicuspidatus	
Diaptomus tyrelli	
Bosmina longirostris	
Leptodora sp.	
Diaphanosoma sp.	
Ceriodaphnia sp.	

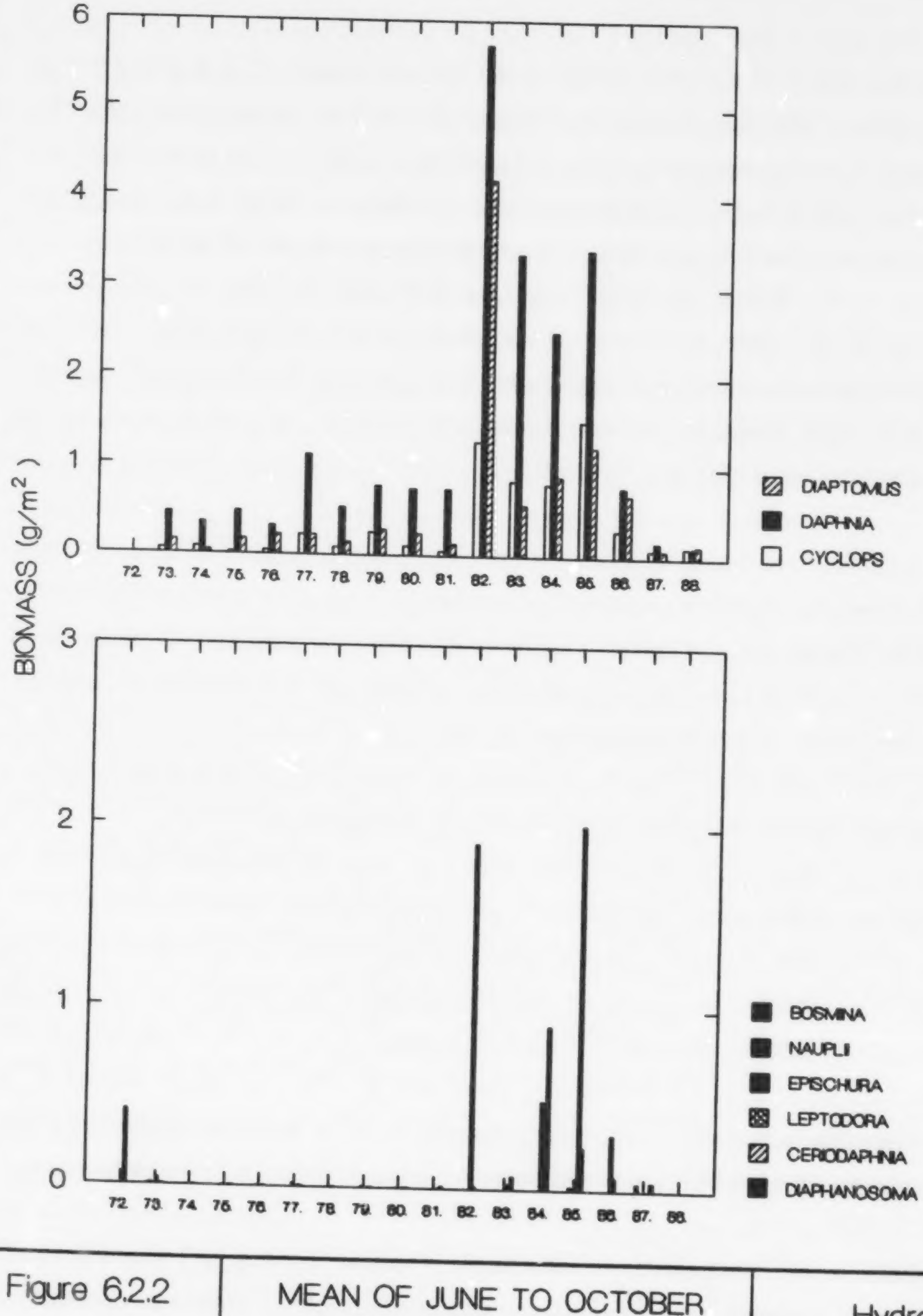

Figure 6.2.2

MEAN OF JUNE TO OCTOBER
ZOOPLANKTON BIOMASS

HydroQual
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Of the rare taxa, the genus that has increased the most since 1982 is Bosmina (Figure 6.2.2). Before 1982, bosminids were present in only 21% of samples; after 1982, they appeared in 39% of samples, and the corresponding mean biomasses were 0.01 and 0.53 g/m^2, respectively. Changes were also apparent for other taxa. Leptodora appeared periodically before 1982; after 1982, however, it did not appear again in samples at the Border site until 1988 (Les McDonald, personal communication). Chisholm et al. (1989) also recorded the disappearance of Leptodora in samples collected just below Kikomun Creek in 1983 and 1984, but they found them again in 1985 and 1986. Therefore, the disappearance of Leptodora roughly coincided with a period of increase in Bosmina numbers.

This reciprocal relationship supports the speculation that Bosmina has increased as a result of reduced predation pressure from Leptodora, in a manner similar to that in Lake Michigan (Branstrator and Lehman, unpublished). We used data from Chisholm et al. (1989) to compare seasonal mean densities (numbers per litre) of these two animals collected in 30-m net hauls at the Kikomun site:

Year	Bosmina	Leptodora
1984	2.06	0.24
1985	0.38	4.62
1986	0.03	4.65
1987	1.50	2.53

Although this is a very small dataset, the significant negative correlation between variables ($r = 0.97$; $p<0.05$) is consistent with the hypothesis of invertebrate predation.

Chisholm et al. (1989) made an alternative proposal that Bosmina had increased because of a release from resource competition with Daphnia, which has declined as a result of size-selective predation by the kokanee. This hypothesis is more difficult to support with present data because there is no apparent correlation between Daphnia and Bosmina densities from 1982 to 1986 (Figure 6.2.2). The fact that Bosmina occurred so rarely before 1982, even though the Daphnia population was relatively small, is also inconsistent with this hypothesis of resource competition (Figure 6.2.1). Further, the poor correlation between zooplankton standing crop and phytoplankton standing crop does not argue for food limitation.

There were changes in other rare genera as well. Epischura was much more common between 1982 and 1986 than previously; it was recorded only once before 1982, but occurred on nine occasions after. Ceriodaphnia and Diaphanosoma were found in low numbers between 1973 and 1982, but were found in relatively large numbers in 1985 (Ceriodaphnia) and 1987 (Diaphanosoma).

6.2.3 Seasonal and Spatial Distribution

Zooplankton tended to be more abundant in late spring (June) and fall than in mid-summer (Figure 6.2.3). This pattern is especially evident in data after 1980. Both Cyclops and Daphnia densities are high in June, and then decline, the latter probably in response to increased predation from kokanee (Chisholm et al., 1989). Diaptomus biomass does not peak until the fall, at a time when Daphnia biomass increases slightly again.

The vertical distribution of zooplankton in the reservoir has been detailed by Chisholm et al. (1989). They found that all species of zooplankton exhibited some degree of variation in their vertical distribution throughout the year. Generally, animals were concentrated at depths above 12 m from May to September, and peak densities could be found between 3 and 6 m. At the Border site, animals have usually been found congregating at 5 m during routine sampling (L. McDonald, personal communication). Between October and March, however, zooplankton were more uniformly distributed

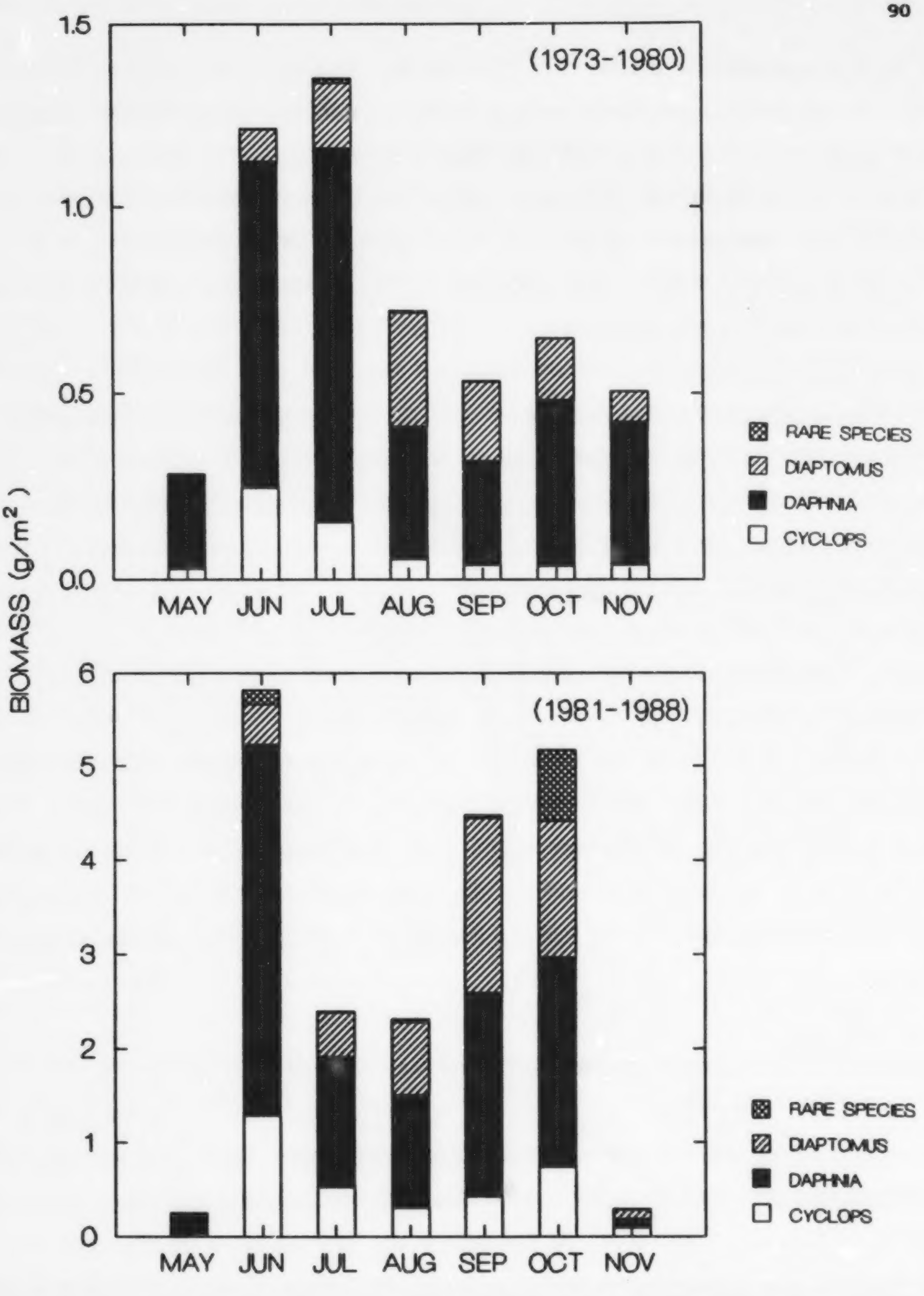

Figure 6.2.3	MONTHLY MEAN ZOOPLANKTON BIOMASS	HydroQual Canada Ltd

in the water column. Daphnia exhibited a slight diel vertical migration within the surface 9 m, in the Rexford area, during the summer of 1986.

There is much less information available regarding the horizontal distribution of animals in the reservoir. We compared densities at the Border site with those from the site near Kikomum Creek (Table 6.2.2). There was a great deal of variation between sites, that varied erratically in both magnitude and direction. Roughly 25% of paired samples differed by an order of magnitude in animal abundance, and 44% of these had densities that differed by at least 300%. This indicates that the distribution of zooplankton on a large scale is extremely heterogeneous within the reservoir. Further studies should be carried out to determine the extent of horizontal patchiness in zooplankton distribution, especially considering the importance of obtaining accurate zooplankton densities in long-term monitoring programs, and the enormous month-to-month variations in zooplankton biomass in 1982-1986 (Section 6.2.1). The efficacy of single monthly net tows at a fixed station needs to be verified.

6.3 Fish Distribution and Abundance

Fishery production in a new reservoir undergoes a characteristic sequence as the waterbody ages. Initially, there is a highly productive period which has been termed "trophic upsurge" (Baranov, 1961). Kimmel and Groeger (1986) coined the term "trophic depression" to describe the period following the initial trophic upsurge, when reservoir productivity declines toward its natural equilibrium level.

The initial period of high productivity results from a number of factors, including:

1. a large influx of organic detritus and inorganic nutrients from the inundated reservoir basin (Kimmel and Groeger, 1986),
2. an abundance of high-quality habitat and food for benthic organisms (Kimmel and Groeger, 1986), and

Table 6.2.2 Comparison of Animal Densities (No./L) from Samples Collected at the Border Site (by B.C. Environment) with those Collected near Site 101 (between Kikomum Creek and Elk River Confluence; by Montana Department of Fish, Wildlife and Parks (Chisólm et al., 1989)).

	Daphnia		Diaptomus		Cyclops		Epischura	
Date	Border	Site 101	Border	Site 101	Border	Site 101	Border	Site 101
1983								
September	1.05	2.81	3.26	4.51	1.13	3.26	0.018	0.016
October	5.08	3.59	8.93	5.09	8.23	4.25	0	0.004
November	1.10	10.47	2.08	7.61	3.67	6.38	0	0.008
1984								
July	0.94	4.85	1.38	2.63	4.65	8.17	0.086	0.009
August	1.18	3.99	0.71	0.83	1.08	2.91	0.021	0.045
September	1.00	0.87	0.59	0.61	1.13	2.20	0.092	0.051
October	52.00	6.51	35.50	2.65	47.00	7.73	1.000	0.022
1985								
June	52.80	4.06	16.00	0.26	51.20	2.90	0	0
July	14.25	4.22	4.60	0.54	11.40	1.14	0	0.003
August	1.89	2.52	5.72	3.08	10.21	3.78	0	0.013
September	7.28	3.99	16.64	14.44	14.04	7.81	0	0.000
October	3.94	4.29	24.75	12.67	20.25	10.03	0	0
1986								
June	2.56	7.65	0.34	4.37	2.84	9.74	0	0.040
July	4.32	2.55	17.46	6.40	19.26	6.77	0	0.008
August	8.10	2.22	15.30	1.71	10.62	3.59	0.360	0.009
September	0.46	3.02	1.17	2.92	1.14	3.75	0.010	0.007

3. a rapidly expanding lacustrine environment (Ploskey, 1982).

The factors identified by Kimmel and Groeger (1986) for the subsequent trophic depression are:

1. decreased internal nutrient loading,
2. a decline in the mass of biologically unstable organic detritus,
3. the cessation of habitat expansion, and
4. a reduction of favourable habitat.

Fish populations which are present during the trophic upsurge and depression stages of reservoir aging may undergo wide fluctuations in numbers, but typically stabilize within 5 to 10 years (Ploskey, 1981; Benson, 1982, Adams et al., 1983).

Lake Koocanusa does not appear to have undergone a true trophic upsurge in the first decade after filling. Populations of zooplankton and phytoplankton (and presumably primary and secondary production) have remained uniformly low during that period (Sections 6.1.1, 6.2.1) and elevated phosphorus and nitrogen levels in the early years are entirely attributable to nutrient income from the tributaries (Section 5.2). Only fish populations demonstrated the pattern of increase and decline anticipated in a newly closed reservoir (Figure 6.3.1). It would be incautious to consider this a trophic upsurge, since only one component of the biotic community, rather than the whole ecosystem, was involved. It also suggests that initial augmentation and subsequent reduction of favourable lacustrine habitat was mostly responsible for the change in fish density, rather than addition of nutrients or organic matter from the basin.

Before the dam was closed, the fishery of the upper Kootenay River consisted primarily of cutthroat trout, rainbow trout, mountain whitefish and burbot. Immediately after impoundment, the trout species and mountain whitefish were common in gill net catches in Lake Koocanusa; in recent years, these species have declined in abundance.

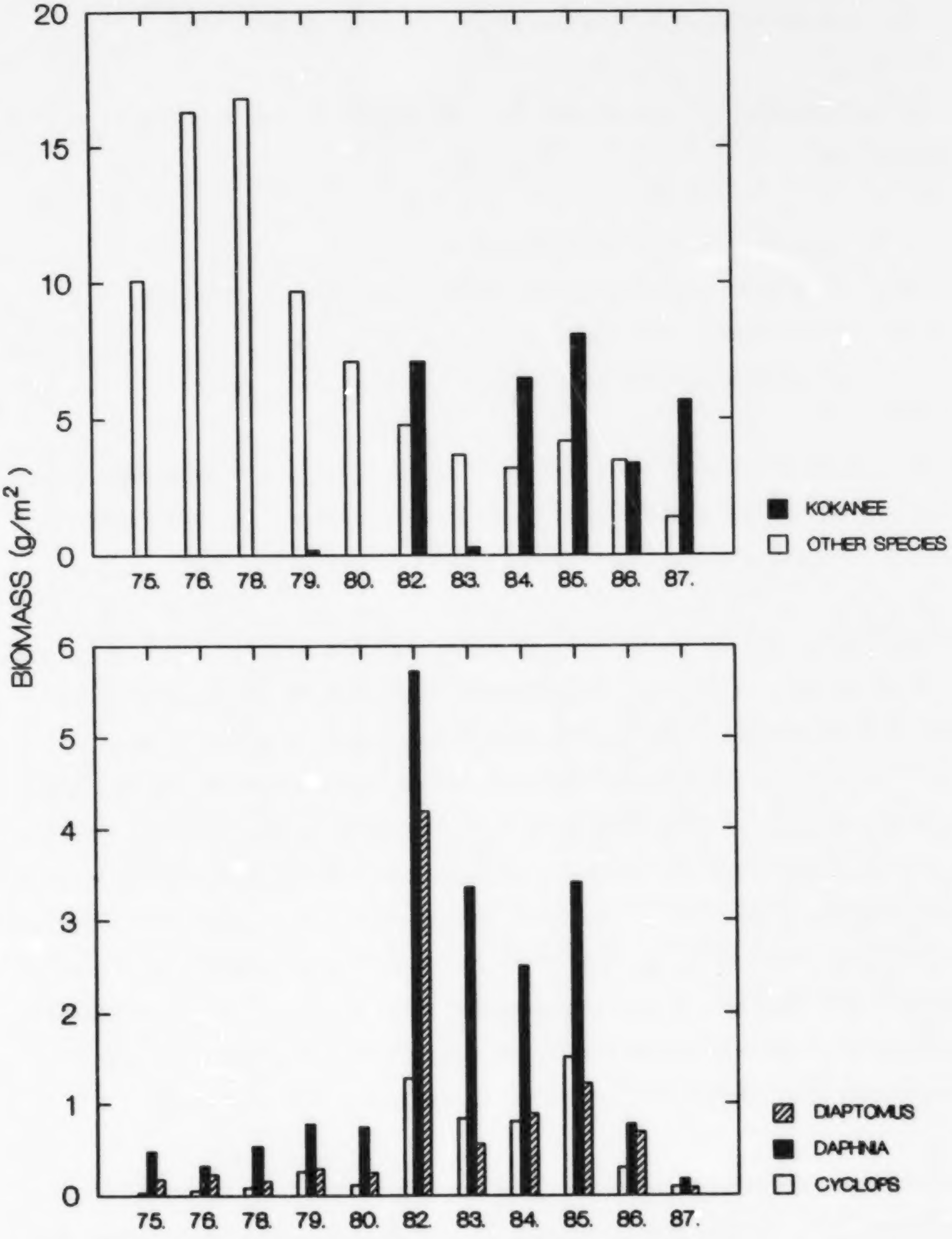

Figure 6.3.1

ANNUAL FISH AND ZOOPLANKTON BIOMASS

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Ten species of game fish and seven nongame species are currently present in the reservoir, of which two (kokanee and yellow perch) are the result of unintentional introductions to the waterbody (Table 6.3.1). The trends in relative abundance of species since 1982 are summarized on Table 6.3.2.

The redside shiner population was initially high after impoundment, but since 1980 it has declined dramatically, possibly because spawning habitat has been lost through extensive shoreline erosion. The mountain whitefish population also declined drastically a few years after impoundment, presumably because of a lack of suitable spawning habitat in tributaries and loss of the spawning substrate within the old Kootenay River channel. The gradual decline in rainbow trout and cutthroat trout since 1978 can be attributed to reductions in planted stock (cutthroat stocking ended in 1982; Huston et al., 1984), limited spawning habitat in tributary streams, and competition with the abundant planktivores in the reservoir. Rainbow trout may also have been reduced because of the decline of its primary fish prey, the redside shiner.

Although the relative abundance of Oncorhynchus trout has declined in the American regions of the reservoir, these species, as well as mountain whitefish, northern squawfish and the Catostomus species presently have a greater relative abundance in Canada. The species assemblage in Canada more closely resembles that of the Kootenay River than the southern regions of Lake Koocanusa, probably because the Canadian portion of the reservoir is most affected by reservoir drawdown. Minimum pool elevations in the last few years have resulted in a dewatering of the Canadian portion of the reservoir (G. Oliver, personal communication). Hence, a lacustrine species assemblage has not replaced the riverine species as it has on the American side of the reservoir.

Chisholm et al. (1989) note a recent increase in the standing stocks of peamouth chub and kokanee in the reservoir; their numerical dominance became apparent in 1982. These authors speculate that the upward trend in the abundance of these species,

Table 6.3.1 Current Relative Abundance of Species Present in Koocanusa Reservoir*

Common Name	Scientific Name	Relative Abundance
Game Fish Species		
Westslope cutthroat trout	*Oncorhynchus clarki lewisi*	Common
Rainbow trout	*Oncorhynchus mykiss*	Abundant
Bull trout	*Salvelinus confluentus*	Common
Brook trout	*Salvelinus fontinalis*	Rare
Lake trout	*Salvelinus namaycush*	Rare
Kokanee salmon	*Oncorhynchus nerka*	Abundant
Mountain whitefish	*Prosopium williamsoni*	Common
Burbot	*Lota lota*	Common
Largemouth bass	*Micropterus salmoides*	Rare
White sturgeon	*Acipenser transmontanus*	Rare
Nongame Fish Species		
Pumpkinseed	*Lepomis gibbosus*	Rare
Yellow perch	*Perca flavescens*	Common
Redside shiner	*Richardsonius balteatus*	Rare
Peamouth	*Mylocheilus caurinus*	Abundant
Northern squawfish	*Ptychocheilus oregonensis*	Abundant
Largescale sucker	*Catostomus macrocheilus*	Abundant
Longnose sucker	*Catostomus catostomus*	Common

* From Chisholm et al., (1989).

Table 6.3.2 Relative Abundance of Fish Species (a) Captured in Vertical Gill Nets in Three Sampling Areas in Koocanusa Reservoir, 1983 Through 1987 (b)

Year	RB	WCT	HB	COMB	DV	KOK	MWF	CRC	NSQ	RSS	CSU	FSU
TENMILE												
1983	7	2	4	13	12	64	1	9	1	0	0	0
1984	1	1	1	3	1	89	<0.1	6	<1	<0.1	<1	0
1985	<1	1	<1	1	<1	89	0	9	<1	0	<1	0
1986	<1	<1	<1	1	1	88	<1	9	0	0	<1	0
1987	<1	<1	1	1	<1	79	<1	19	<1	0	<1	0
REXFORD												
1983	4	4	0	8	11	39	0	22	20	0	0	0
1984	2	1	1	4	1	85	1	7	1	<1	1	<1
1985	1	<1	<1	1	<1	57	<1	37	1	<1	<1	0
1986	2	0	1	3	<1	47	2	44	2	0	1	0
1987	<1	<1	0	1	1	72	<1	25	<1	<1	0	0
CANADA												
1984	2	0	2	4	0	14	3	44	14	0	17	3
1985	<1	<1	0	1	<1	24	0	67	6	1	0	0
1986	1	2	1	5	1	5	2	80	3	0	4	<1
1987	2	1	<1	3	0	9	1	82	4	<1	<1	0

RB = rainbow trout, WCT = westslope cutthroat trout; HB = rainbow x cutthroat trout hybrid, COMB = combined trout, DV = bull trout, KOK = kokanee salmon, MWF = mountain whitefish, CRC = peamouth chub, NSQ = northern squawfish, RSS = redside shiner, CUS = larger scale sucker, FSU = longnose sucker.

From Chisholm et al. (1989).

along with yellow perch, will continue but ultimately will be controlled by competition for plankton in the reservoir.

The direct relationship between zooplankton production and standing crops of fish species which use that production is, however, confounded by a number of factors related to the development and behaviour of the fish species involved. For example, Kokanee was first detected in lake Koocanusa in 1979, when a few two-year-olds were collected during routine sampling. The origin of these fish was uncertain. The majority of the kokanee stock in the reservoir originated from the accidental release of 250,000 fry from the Kootenay Trout Hatchery in 1980, which resulted in a very strong year-class of spawners in 1982, and at three-year intervals since that time. Fluctuations in the abundance of this species will continue according to the strength of various year-classes.

Daphnia are by far the most important food item for kokanee in Lake Koocanusa (Table 6.3.3), as is the case in other lakes. The copepod Diaptomus is the second most important food for this species. Chisholm et al. (1989) showed that the number of large (i.e. 1.5-1.9 mm size class) Daphnia taken by kokanee showed an inverse relationship to the population density of kokanee in the reservoir, i.e., the fish compete for the large Daphnia. Size selectivity appeared to change with the seasons, with selection for larger individuals increasing during the winter through summer period and decreasing in the fall. The implication is that the kokanee are exerting size-selective pressure on the Daphnia population.

The apparent six-fold increase in biomass of Daphnia from 1981 to 1982 coincided with the appearance of kokanee in the reservoir (Figure 6.3.1). According to the hypothesis of cascading trophic interactions (Carpenter et al., 1985), when the biomass of a size-selective vertebrate planktivore (such as the kokanee) increases, the biomass of large herbivorous zooplankton should decrease, while that of small herbivores should increase. Although the small herbivore Bosmina increased in 1984, the larger forms

Table 6.3.3 Indices of Relative Importance for Food Items in the Stomachs of Kokanee Salmon in Koocanusa Reservoir, 1983 Through 1987.*

Sample Size	N = 26 1983	N = 118 1984	N = 129 1985	N = 84 1986	N = 29 1987
ZOOPLANKTON					
Daphnia	83	74	59	96	72
Diaptomus	10	19	29	2	29
Epischura	3	5	6		
Other plankton		0			
TERRESTRIAL INSECTS					
Hymenoptera		0	0		
Cleoptera	1	1			
Hemiptera					
Other insects		0	1		
AQUATIC INSECTS					
Diptera larvae			0	1	
Diptera pupae	1	0	3	0	8
Insect parts	1	1	1		
Debris		0			

From Chisholm et al. (1989).

such as Daphnia and Diaptomus also increased. Therefore, the increased biomass of these large crustaceans in 1982 cannot be directly attributed to the presence of the kokanee.

However, kokanee may still have been indirectly responsible for the zooplankton surge for the following reasons. First, the zooplankton increase did not correspond to any consistent change in nutrient concentrations or in standing stock or community structure of phytoplankton. Therefore, a top-down control (fish predation) rather than a bottom-up control (food availability) must have been responsible. Second, Kokanee feeding strongly favours Daphnia. Therefore, while they may have caused the declining relative dominance of Daphnia, they cannot have caused the observed increase in their absolute biomass. Third, the zooplankton increase began when the Kokanee fry released in 1980 reached breeding age, and persisted for exactly the length of an average life cycle (Scott and Crossman, 1973). It is reasonable to expect a connection between the two events. Finally, a number of other fish species, including trout and squawfish at least, feed on large plankton as well as juvenile kokanee. Therefore, it is hypothesized that when the introduced kokanee spawned in 1982, they provided an abundant food source, kokanee fry, that was strongly preferred by other omnivorous fish species. This relieved predation pressure on the zooplankton population, normally kept in check by planktivores, and produced the observed increase in zooplankton biomass. As the stock of young kokanee was depleted, and the growing fish themselves began consuming significant quantities of plankton, the population surge would decline as observed. Such indirect effects of fish population changes have been observed in other lakes.

If this hypothesis, or a variant of it, were correct, then there are two present possibilities: either the ecosystem has reached a new equilibrium and zooplankton populations will remain stable, or there will be a similar, probably smaller biomass increase about 3-4 years after the last one, i.e. in 1989 or 1990. Hence, zooplankton and fish populations warrant careful scrutiny.

The kokanee population in Lake Koocanusa may also be expected to undergo seasonal movements, both as a result of the temperature preference of this species and its reproductive activities. Chisholm and Fraley (1985) noted that during the summer months, kokanee were found almost exclusively at depth strata where optimum temperatures (i.e. 10-15°C) prevailed, and when temperatures were below 10°C, kokanee distributions were more strongly influenced by Daphnia abundance. The abundance of kokanee in Canada may be expected to decrease, regardless of zooplankton densities, as reservoir warming occurs in the summer months, as a result of movements to the deeper regions of the reservoir in the United States where preferred temperatures exist. In the fall, relative abundance of kokanee probably increases in the Canadian area of the reservoir, as spawners return to their natal streams. This may cause some of the seasonal variation in zooplankton density at the Border site.

The relationship between zooplankton productivity and pelagic consumers also has implications for several other sport fish in Lake Kookanusa. Daphnia were the most important zooplankton taken by both kokanee and mountain whitefish. In addition, Daphnia were the most important zooplankton taken by largescale sucker, although this species also preyed heavily on Cyclops which were not pursued by either the kokanee or mountain whitefish. However, because of behavioural differences, little competition may exist between these species in Lake Koocanusa (Chisholm et al., 1989). Largescale suckers and mountain whitefish commonly occupy near-shore areas close to the substrate, while kokanee are primarily pelagic.

There is also an overlap in the zooplankton preferences of kokanee and Oncorhynchus trout during the winter months. Whereas trout fed more on aquatic and terrestrial invertebrates in the spring and summer, and on other fish (notably peamouth chub, redside shiner and northern squawfish) in the fall, both trout and kokanee diets consisted primarily of zooplankton in the winter. However, Chisholm et al. (1989) suggested that intense competition between kokanee and the trout species may not occur during winter because the mean lengths of Daphnia ingested by trout were considerably greater than those taken by kokanee (1.70 mm and 1.48 mm, respectively). Trout continued to feed on aquatic and to a lesser extent terrestrial invertebrates, while kokanee utilized Diaptomus as an important secondary food source between December and March.

In summary, since TP has generally decreased in the reservoir (see Section 5.2), and the overall phytoplankton community has not changed over the fifteen years of record (as indicated by chlorophyll a and phytoplankton biomass; Figure 6.1.1), it is unlikely that the sudden increase of zooplankton in 1982 can be explained by changes that occurred in the base of the food-web. An indirect effect, by kokanee fry diverting mixed predators, is a possible cause. The change in the zooplankton community with respect to the proportions of Daphnia, Cyclops and Bosmina since 1982 are probably the result of predation from various invertebrate and vertebrate planktivores, especially selectivity for Daphnia by kokanee. We cannot, however, be certain whether the biomass estimates of zooplankton truly reflect an overall increase in animal abundances throughout the reservoir (because of supposed horizontal heterogeneity in the zooplankton distribution, and the limitations of data from a single site; see Section 6.2).

7.0 TROPHIC STATUS OF THE RESERVOIR

Trophic state pertains to the general biological status of a waterbody. An oligotrophic ecosystem has low algal, zooplankton and fish productivity, and usually has high water clarity. In eutrophic lakes algal growth is abundant and severe algal blooms frequently occur. Blue-green algae are usually conspicuous, and water clarity is often low. Fish production can be high in eutrophic waterbodies, unless production is limited by low oxygen levels. Mesotrophy is a trophic category intermediate between oligotrophy and eutrophy. In most lakes and reservoirs trophic state is controlled by the ecosystem's nutrient content.

Summary statistics of trophic indicators for Lake Koocanusa are presented in Table 7.1.1. (Some of the older entries in the database do not specify a detection limit; this results in minima of zero for some parameters.) Both TP and chlorophyll a concentrations were lower at the Forebay site than at the Border site, and the shift in phosphorus content after 1977 is evident (see Sections 5.2 and 6.1).

Figure 7.1 compares total phosphorus and chlorophyll a concentrations in the reservoir according to the schema for British Columbia lakes developed by Nordin and McKean (1984). Although there is substantial year-to-year variability, summer average chlorophyll a concentrations at the Border site were within the broad range predicted from spring concentrations of total phosphorus, both before (except 1 datum) and after 1976, when tributary TP loads were reduced. At the Forebay site, chlorophyll a levels tend to be on the low side of the range expected for the corresponding TP level, on three occasions (after 1977) even exceeding the lower 95% confidence limit (Figure 7.1). The limited data suggest that the noncorrespondence may have been greater before 1977, when TP loadings were substantially greater. These results substantiate the suggestion that TP is not always limiting primary production in Lake Koocanusa (Woods and Falter, 1982).

TABLE 7.1.1 Summary Statistics of Lake Koocanusa Trophic Indicators for Border and Forebay Euphotic Zone (top 10 m). Note that Forebay Data contained zero values for the Total Phosphorus and Chlorophyll a that should have been reported as less than detection limit (0.001 for TP)

SITE		N OF CASE	MINIMUM	MAXIMUM	MEAN	STANDARD DEVIATION
BORDER	TOTAL PHOSPHORUS (mg/l)	before 1977 86	0.004	0.540	0.045	0.065
		1977 & after 165	0.003	0.039	0.011	0.006
	CHLOROPHYLL a (ug/l)	197	0.700	9.000	2.030	1.270
	SECCHI DEPTH (m)	182	0.457	10.000	4.019	2.446
	TOTAL NITROGEN (mg/l)	197	0.030	0.610	0.185	0.098
	TN : TP RATIOS	198	0.650	102.000	20.082	14.889
FOREBAY	TOTAL PHOSPHORUS (mg/l)	before 1977 141	0.010	0.100	0.035	0.023
		1977 & after 286	0.000	0.061	0.009	0.008
	CHLOROPHYLL a (ug/l)	569	0.000	7.780	1.260	1.480
	SECCHI DEPTH (m)	N/A	N/A	N/A	N/A	N/A
	TOTAL NITROGEN (mg/l)	369	0.010	2.500	0.441	0.318
	TN : TP RATIOS	253	1.000	195.000	54.457	43.917

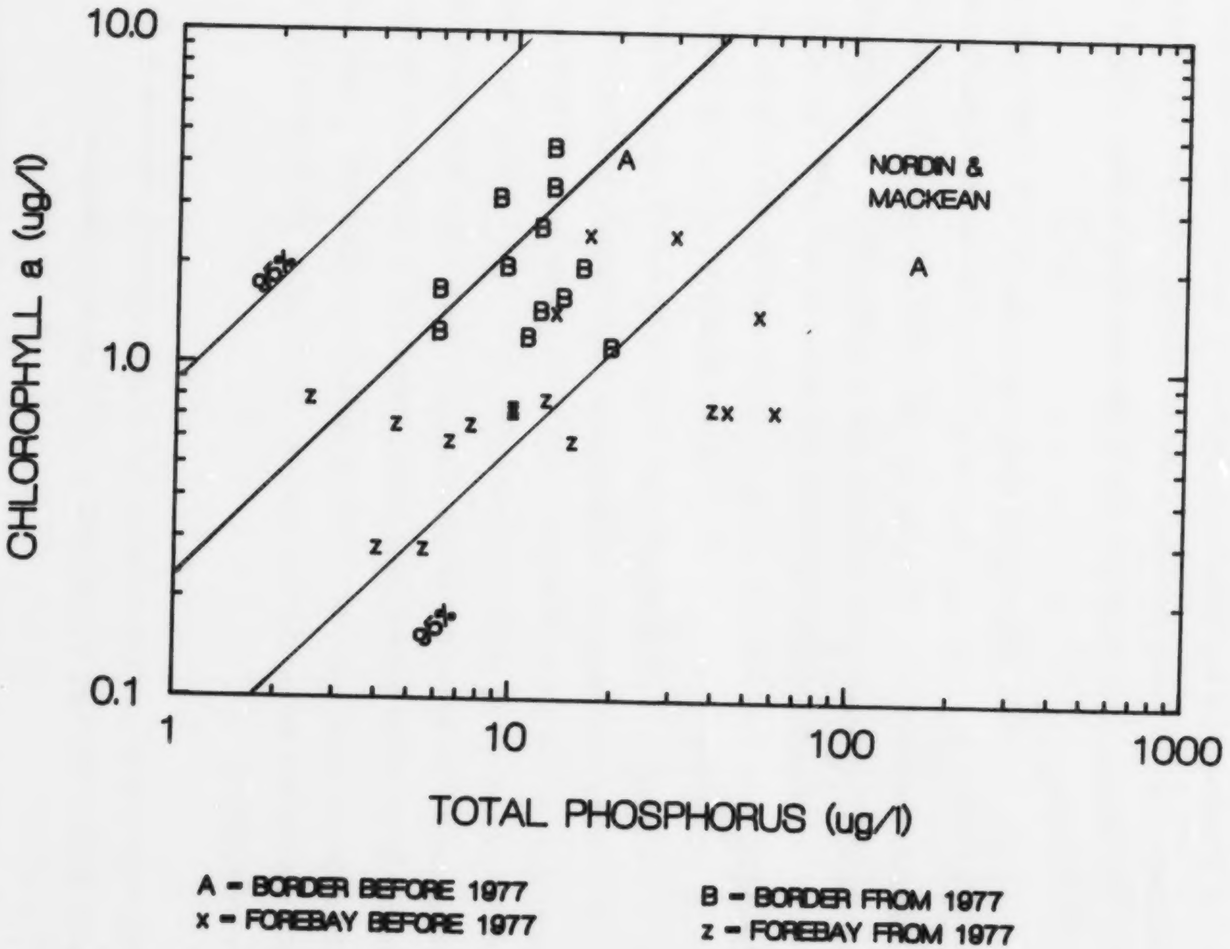

Figure 7.1

KOOCANUSA BORDER SITE
PHOSPHORUS CHLOROPHYLL a RELATIONSHIP

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Carlson (1977) formulated a trio of trophic state indices (TSI) to aid lake managers in evaluating the trophic condition of lakes, and to communicate these findings to the public. The indices rely on empirical relationships among Secchi depth, chlorophyll a and TP concentrations and are expressed as a productivity value between zero (lowest) and 100 (greatest). The TSI values associated with TP, chlorophyll a and Secchi depth from 1972 to 1989 are plotted separately for the Border and Forebay sites in Figure 7.2.

Before treatment of Cominco's fertilizer plant discharge began in 1975, there were high concentrations of TP in the reservoir, which should have led to much higher algal growth, if that had been the sole controlling variable. After the abatement measures were implemented, TP concentrations declined, and subsequent TSI values associated with TP were also reduced to levels consistent with those of chlorophyll a and Secchi depth. After 1976, there was generally close agreement between the phosphorus and chlorophyll TSI values, with both fluctuating between 30 and 40 units. This places the Border site in the upper end of the oligotrophic range. The Secchi depth TSI borders on mesotrophic, probably in response to the high non-algal turbidity experienced at the Border site in May and June. At the southern end of the reservoir, the phosphorus and chlorophyll TSI indices are in close agreement after 1977. The Forebay index values are somewhat lower than at the Border site, placing it clearly in the oligotrophic range. The constancy of the three trophic state indices from 1977 to 1989 confirms that the base of the reservoir food web has remained stable throughout this 14-year period. Before 1977 there was a strong disparity between TSI based on phosphorus and those based on chlorophyll a.

Woods and Falter (1982) concluded that the lack of phytoplankton response to the high phosphorus levels in the early years (1972 to 1975) could be attributed to:

- weak thermal structure which allows phytoplankton to circulate out of the euphotic zone;

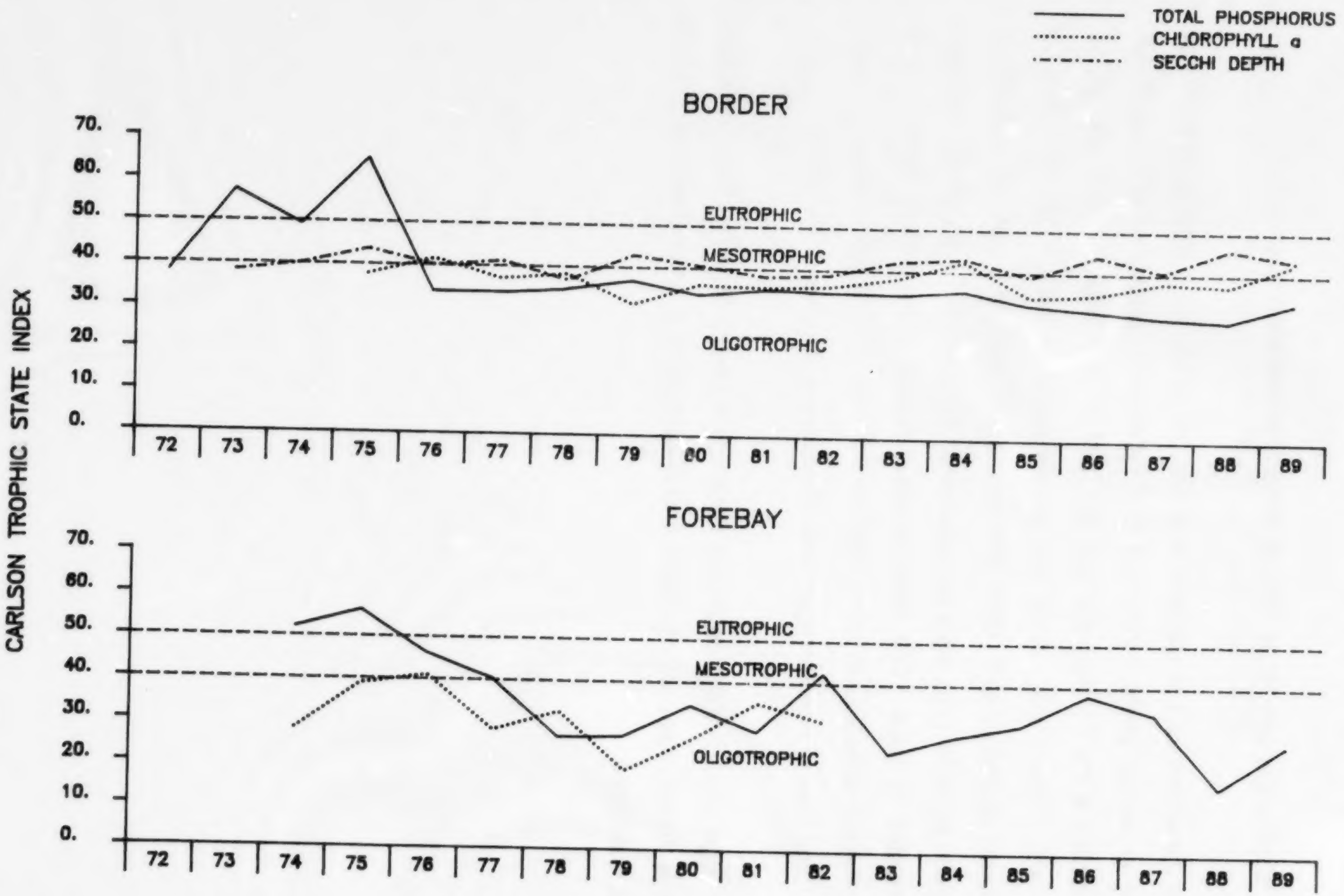

Figure 7.2

CARLSON TROPHIC STATE INDICES

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- light inhibition created by highly turbid inflows from the Kootenay River;
- short water residence time causing phytoplankton washout.

In addition to these factors it is possible that nitrogen could be a limiting factor either seasonally or in certain years. Very low total nitrogen to phosphorus ratios (Table 7.1.2) were evident at the Forebay site over the 1974 to 1977 interval, and at the Border location in 1975. These low values encompass many of the datapoints which fall outside of Nordin and McKean's (1984) 95% confidence interval (Figure 7.1). Nitrogen fixing blue-green algae blooms were also evident in the fall of 1974 and 1975, and algal assays conducted in June 1973 were nitrogen limited (Woods and Falter, 1982). Nutrient enrichment experiments conducted on the U.S. side of the reservoir in 1987, indicated co-limitation of both nitrogen and phosphorus (Dodds et al., 1989).

Based upon algal abundance, Koocanusa Reservoir is clearly oligotrophic. The factors regulating this trophic state are complex, and likely include a combination of physical and nutrient conditions.

Table 7.1.2 Median Euphotic Zone (top 10 m) TN:TP Ratios for Border and Forebay Sites.

Year	Border	Forebay
1974	20	7
1975	3	4
1976	13	9
1977	21	9
1978	17	80
1979	22	44
1980	48	80
1981	12	76
1982	18	76
1983	12	128
1984	16	80
1985	20	60
1986	20	44
1987	23	60
1988	22	120

8.0 SUITABILITY FOR USE

We wished to test water quality of Lake Koocanusa and the Kootenay and Elk Rivers with respect to end uses. No water quality objectives, however, have been established specifically for these waterbodies. We therefore reviewed water quality criteria published by the British Columbia Ministry of Environment and Parks, and selected the appropriate criteria (Table 8.1.1) for this review. The criteria selected assume the primary uses of water in the reservoir to be 1) public water supply, 2) supporting aquatic life, 3) irrigation and 4) recreation, and are based upon the most sensitive of these uses in each case.

It should be noted that the B.C. water quality criteria are intended to be used as a screening tool (Pommen 1989) and it is in this fashion that the values in Table 8.1.1 are being tested; they should not be construed as water quality objectives. If the water body meets the criteria there are unlikely to be problems. If exceedances are detected then further evaluation of the data, and possibly the development of site-specific objectives, are warranted.

8.1 Explanation of Criteria

In most cases the criteria we used were obtained directly from water quality criteria publications. However, the toxicity of certain metals (cadmium, copper, lead, nickel) decreases with increasing water hardness, so we based the suggested criterion for these metals upon the historical hardness value that was exceeded 75% of the time, thereby providing a rather stringent value against which to test (75% of the time we would be overprotective if our criteria were used as fixed water quality objectives). Similarly, the toxicity of ammonia increases with pH and temperature, so we based our criterion value upon the pH and temperature readings that were exceeded only 25% of the time in the historical data. For the reservoir, data from only the euphotic zone (to 10 m) were used.

Table 8.1.1 Water Quality Criteria Tested for Koocanusa Reservoir, Kootenay and Elk River

	Koocanusa Reservoir	Kootenay & Elk Rivers	Determining Use
Total Arsenic (ug/L)	50	50	AL,PWS
Total Cadmium (ug/L)**	0.8	0.8(1.3)*	AL
Total Copper (ug/L)**	4	4.2(5.2)*	AL
Chlorophyll a	2.5 ug/L	50 mg/m2	PWS,R
Total Chromium (ug/L)	2	2	AL
Coliforms Fecal (#/100 ml)***	10	10	PWS
Colour (True or TAC)	15	15	PWS,R
Fluoride (mg/L)	0.3	0.3	PWS,I,AL
Total Iron (mg/L)	0.3	0.3	PWS,AL
Total Lead (ug/L)	6.5	6.5(7.75)*	AL
Total Manganese (ug/L)	50	50	PWS
Total Nickel (ug/L)**	65	65(110)	AL
Total Ammonia (mg/L)	0.424	0.562	AL
Nitrate (mg/L)	10	10	PWS
Oxygen (mg/L)	8	8	AL
pH (pH units)	6.5–8.5	6.5–8.5	PWS
Total Phosphorus (ug/L)	10		PWS,R
Total Zinc (mg/L)	0.014	0.03	AL

* Value in parentheses applies to Elk River

** Based on Hardness of 100 mg/L for Koocanusa and Kootenay Rivers, 130 mg/L for Elk River

*** 30 day 90th percentile

NOTE: All guidelines are derived from criteria reported in: Pommen, L.W. 1989. Approved and Working Criteria for Water Quality. B.C. Ministry of Environment, Water Management Branch, Resource Quality Section, except for Chromium in Rivers CCREM and Fluoride which is the tentative MOE criterion (L. Pommen, Pers, Comm).

AL – Aquatic Life	PWS – Public Water Supply
R – Recreation	I – Irrigation

The criteria documents suggest a level of 11 mg/L for oxygen for the protection of fish embryos and larvae. Strictly speaking, this criterion need only be applied to the sites and seasons where these life stages are found, not necessarily the entire system. We therefore chose the lower value of 8 mg/L, which is a general criterion for salmonids. The criteria documents also indicate that water quality criteria for ammonia, copper and lead should be evaluated against thirty-day averages based upon at least five, approximately weekly samples. We did not have sufficient data for such a test so, as with other variables, we compared the criteria with spot observations.

8.2 Testing the Criteria

A number of the criteria listed in Table 8.1.1 were tested for exceedance in the Kootenay and Elk Rivers and in Lake Koocanusa at the Border site (Table 8.2.1). The reasons for choosing the constituents to be tested were their perceived importance, and the availability of a sufficiently large number of observations in the database. Readings in the data base associated with techniques whose detection limits were greater than the criterion of interest were removed before testing.

There were relatively few exceedances of the criteria. Some of the metals did show exceedances, but these appear to be associated with either: 1) naturally occurring levels, 2) contamination from Cominco Kimberley Operations in the 1970's, which is now rectified, or 3) analytic problems.

Chromium

Suitable data for testing the chromium criterion were only available for 1987 and 1988. Levels of this metal frequently exceeded the selected criterion. It appears, however, that since exceedances were more or less equally common in both rivers, that the exceedances may have resulted from naturally occurring chromium in suspended

Table 8.2.1 Frequency of Exceedances of Water Quality Guidelines by Historic Data. Numbering exceeding: Number complying in Lake Koocanusa (Border), Kootenay River and Elk River. For data 1972–1989 except Chromium which was 1987–1988

		SITE	
Variable	**Koocanusa**	**Kootenay**	**Elk**
Arsenic	0:24	0:118	0:78
Cadmium	NA*	2:138	0:103
Copper	37:125	80:209	61:169
Chromium	NA*	17:14	20:7
Faecal Coliforms	6:35	30:9	13:2
Colour	4:183	41:193	9:184
Fluoride	12:26	63:234	1:249
Iron	30:171	136:132	59:184
Lead	2:128	29:239	3:203
Manganese	1:145	43:228	12:203
Nickel	0:78	0:147	0:90
Ammonia	0:251	2:318	0:326
Nitrate+Nitrite	0:251	0:320	0:326
Oxygen	172:476	6:136	1:146
pH**	0:507:39	0:420:3	1:406:35
Total Phosphorus	126:125	NA	NA
Zinc	2:131	66:219	7:221

* All observations at border with detection limit greater than objective

** The three frequencies respectively indicate the number of observations below 6.5; the number in the range 6.5 to 8.5; and the number exceeding 8.5

sediments.

Copper

In 1983-1984 and again in 1987-1988 there were numerous copper exceedances (roughly a third of all samples), at all three sites. As discussed in Section 4.1, these exceedances arose from a known contamination problem in the sample bottles, and do not reflect true copper levels in the basin.

Faecal Coliforms

All coliform exceedances occurred before 1977 in the Kootenay River, and before 1976 in the Elk River. In the Kootenay River the end of the exceedances coincides with changes in operation of the Cranbrook sewage treatment plant (Table 1.3.2). At the Border site, in the reservoir, one exceedance occurred in 1986, and the remainder in 1973 and 1975.

Fluoride

As indicated in Section 4.3, there was a marked decrease in fluoride concentration in the Kootenay River in 1975, associated with improved wastewater treatment at the Cominco fertilizer plant. Of the exceedances reported in Table 8.2.1, all but five in the Kootenay River occurred in 1975 or earlier.

Iron

The criterion for total iron of 0.3 mg/L is frequently exceeded in B.C. rivers from the natural iron content of suspended sediments (L. Pommen, personal communication). The temporal pattern of exceedances in the Kootenay River follows a downward trend similar to that for concentrations described in Section 4.1. This is consistent with the commissioning of the drainage water treatment plant at the Cominco Mine. At the other two sites there did not appear to be any temporal pattern in frequency of exceedance.

Lead

Over the period 1973 to 1989 there is a statistically significant decrease in the proportion of exceedances on the Kootenay River (chi square=41.3, $p<0.05$). At the beginning of the record about one third of the samples exceeded the criterion, while by the late 1980's this had dropped to about 4%. The time-series analysis (Section 4.1) was not able to detect a long-term declining trend in the medians of these data; hence the present result suggests that there is a long-term declining trend in the annual maximum values. This trend may also have resulted from the decrease in metal output from the Cominco Mine.

Manganese

Like iron, manganese exceedances in B.C. rivers are fairly common. In the Kootenay River there has been a significant decrease in the frequency of exceedance since 1973 (chi square = 77.93, $p<0.05$). Indeed between 1983 and 1989, five of the seven years showed no exceedances, despite a doubling in sampling effort since 1985. The major decrease in exceedances occurred between 1980 and 1983, shortly after the Cominco Drainage water treatment plant went into operation.

Dissolved Oxygen

Dissolved oxygen only rarely dropped below the 8 mg/L limit in the Kootenay River, and only did so once in the Elk River. There was a higher frequency of violation of the criterion in Lake Koocanusa, however the data tested were from the entire depth profile, so most of the violations reflect reduced dissolved oxygen with depth (Figure 3.3.1).

Total Phosphorus

The total phosphorus criteria was based upon protection of public water supplies and recreation. Since no public water supplies are drawn from PWS users of Lake Koocanusa, it is the recreational resource which is most critical. The criterion was designed to protect the aesthetic attractiveness of the water body with respect to water clarity. It does not appear that water clarity in Lake Koocanusa is primarily regulated by phytoplankton, so the exceedances are of little concern. The criteria for salmonid management is 15 μg/L which is greater than that for recreation and water supply. In recent years this concentration would not have been exceeded.

Zinc

Fifty-five of the sixty-six exceedances occurred before June 18, 1979 when the Cominco drainage water treatment began. Between November 1986 and July 1987, seven of the remaining eleven exceedances occurred. It is not clear whether these were due to some activity in the basin or to a laboratory problem. The latter is suspected.

8.3 Summary

In the historical data, the Kootenay River demonstrated the largest number of criteria exceedances, many of which appear to have been associated with industrial or municipal discharges. The implementation of drainage water treatment at the Cominco-Sullivan mine appears to have had a large mitigative impact such that many of the metals that frequently exceeded our criteria before 1979 now do so only rarely.

Although the criteria for iron, manganese, dissolved oxygen and phosphorus are exceeded, this does not pose a significant limitation to water use. The criteria for these parameters should be reviewed and possibly adjusted if and when water quality objectives are developed for Koocanusa Reservoir and its tributary rivers.

9.0 SUMMARY AND CONCLUSIONS

Lake Koocanusa is a long, narrow reservoir, straddling the international border between British Columbia and Montana. It was impounded in 1972 and since 1974 has undergone very predictable annual fluctuations in lake surface elevations, which are controlled by spring inflow volumes from its major tributaries (the Kootenay, Elk and Bull Rivers), and the rate of drawdown in the fall (for hydroelectric power generation). In a year of average runoff, the reservoir is filled by August and drawdown commences in November and continues until April the following year.

Because of basin morphometry, continual movement of water from reservoir operations and exposure to strong prevailing winds, the Canadian portion of Lake Koocanusa remains mixed from May to November; it undergoes weak thermal stratification periodically. Further downstream at the Forebay site, where maximum depth exceed 100 m, there is a more persistent metalimnion and a steeper thermocline from June to September. The reservoir tends to be well-oxygenated during most of the open-water season and DO generally stays above 7.0 mg/L in the upper 25 m.

The reservoir experiences extremely low Secchi depth values (<0.5 m) in spring, which reflect high sediment loading from the tributaries during spring runoff. Secchi depth is negatively correlated with turbidity and concentration of non-filterable residue), indicating that light penetration in the reservoir during the early spring is greatly influenced by the presence of suspended sediment.

In the Kootenay River, concentrations of total iron and total zinc have declined since 1973, with the largest decrease occurring in 1979. This decline can be attributed to the commissioning of Cominco's drainage water treatment plant in October 1979, which treated mine and tailings pond discharges. Similar decreases are evident in concentrations of total lead and total manganese, but not copper, which has never been discharged at high concentrations. Concentrations of all metals may have increased again in 1985-1987, but the increase was temporary, and probably an artifact of sampling problems.

Concentrations of metals in the Elk River are always lower than in the Kootenay River. There has been an apparent long-term decrease in total iron concentration in the Elk River, although this is probably an analytical artifact. No long-term changes could be detected in concentrations of total copper, total manganese, total zinc or total lead in the Elk River.

Both total phosphorus and soluble reactive phosphorus concentrations have declined significantly in the Kootenay River; the biggest single reduction coincided with the commencement of Cominco's pollution abatement measures in 1975, when supernatant from the fertilizer plant tailings pond was recycled as cooling water rather than discharged into the river. In the Elk River, TP concentrations have not changed significantly over the past 18 years. Although Elk River SRP concentrations have declined since 1968, this long-term trend is very slight compared with that in the Kootenay River and is probably an artifact of changing analytical techniques.

Ammonia and organic nitrogen (total Kjeldahl nitrogen) have also decreased significantly in the Kootenay River since the early 1970's. Most of the decline in ammonia occurred in 1974-1976 and resulted from improvements in waste treatment at Cominco's fertilizer plant. In contrast to the Kootenay River, nitrite-nitrate concentrations in the Elk River have increased consistently since 1971 because of residuals from nitrate-based explosives used in coal mining.

Generally, colour values in the Kootenay River exceeded those in the Elk River, reflecting effluent loading from Crestbrook Forest Industries kraft pulp mill near Skookumchuck. Colour in the Kootenay River has declined since 1981 primarily because of installation of rapid infiltration basins to treat the pulp mill effluent.

Concentrations of total dissolved solids, which tend to be dominated by naturally occurring ions, have not changed significantly in either the Kootenay or Elk Rivers over the past two decades. Concentrations of suspended matter have not changed in the Kootenay River, while they have decreased in the Elk River. The reduction in the Elk River may be attributed to efforts expended by coal mines in the upper Elk River drainage basin to control sediment release from their sites, especially during spring runoff. Fluoride concentrations in both the Elk and in particular, the Kootenay River have been significantly reduced since 1975. The decrease in the Kootenay River reflects improved treatment of wastewaters at the Cominco fertilizer complex. The decrease in the Elk River has been steady over the 18 years of record; concentrations in that river are an order of magnitude less than levels (before 1975) in the Kootenay River.

None of the reservoir concentrations of arsenic and cadmium were above the detection limit during the period of study, and less than 15% of the zinc samples had measurable quantities. By comparison, iron, manganese, copper and lead were present at detectable concentrations in a greater proportion of the samples, although mean concentrations were much lower than those in the two major tributaries.

Reductions in reservoir SRP concentrations at the International Border and the Forebay site paralleled those observed in the Kootenay and Elk Rivers. Generally, the more recent trend is toward a more even distribution of phosphorus with depth at the Forebay site, and for an accumulation of phosphorus near the sediment at the Border. This tendency for phosphorus to accumulate at the bottom in the Canadian portion of the reservoir indicates that Lake Koocanusa acts as a phosphorus sink, and explains why concentrations at Forebay were generally lower than those at the Border site.

There were significant long-term decreases in TN, TKN and NH_3 concentrations at the Border site which probably resulted from reduced Kootenay River loadings. By comparison, there were slow but steady increases in both TN and TKN further downstream at the Forebay site; the reason for which is unknown. There is no evidence that increased concentrations of nitrite-nitrate from the Elk River have had any significant effect on ambient reservoir concentrations.

An inter-agency comparison of date collected at the Border site between 1978 and 1989 revealed no discrepancies between U.S.G.S. and B.C. Environment with respect to TP, SRP, and nitrate-nitrite concentrations; however, concentrations of TN, TKN and NH_3 measured by the U.S. agency were statistically higher than those measured by the Canadian agency. The deviation is greatest for TKN.

Evidence for phosphorus limitation of algal productivity is equivocal; TN:TP ratios at the Border site are >12 only in midsummer, and at the Forebay site interpretation is complicated by the discrepancy between data sources and the extreme variance of the TN:TP ratios.

The pH of water in the reservoir has increased significantly from 1972 to 1989, although the total increase was only 0.2 pH units over the 18 years of monitoring, and may reflect higher pH in the Kootenay River, after cessation of acid mine drainage to the St. Mary River tributary in 1979. TDS concentrations measured in the 1980's were slightly lower than those measured between 1972 and 1977. There has been no significant long-term change in either alkalinity or specific conductance in the reservoir.

Algal standing stock in the reservoir, as indicated by chlorophyll a and phytoplankton biomass, has varied without any significant trend from 1973 to 1978, although correspondence between the two variables was poor during the first four years of reservoir operation. Annual mean concentrations of chlorophyll a tended to range between 1.0 and 2.0 μg/L throughout the period of study, except in 1976 and 1984 when it approached 3.0 μg/L.

Despite the large year-to-year variations in the percent composition of major algal groups in the reservoir, the assemblage is best described as a diatom-chrysophycean community. Dominant diatom taxa include both pennate forms such as Tabellaria fenestrata, Fragilaria crotinensis and Asterionella formosa, as well as centric forms such as Melosira sp. and Cyclotella sp. The dominant chrysophyte genus is Dinobryon, and various species are represented. These two groups alternate in dominance with dinoflagellates, especially Ceratium, and to a lesser extent with ensheathed colonial blue-green algae such as Gleocystis, Choorococcus and Aphanothace. Filamentous blue-greens such as Anabaena and Aphanizomenon were also prominent immediately after impoundment (1974 and 1975).

Algal productivity, both in terms of standing stock and production rates, peaks in mid-summer (July) when surface temperatures are highest and the euphotic zone extends to its greatest depth. Primary productivity in the upper reservoir may be limited by light, especially in May and June; Chlorophyll a levels at the Forebay site tend to be less than expected for a given level of TP. Factors controlling algal productivity are complex and likely include both physical and nutrient parameters.

Before 1982, the mean seasonal biomass of zooplankton at the Border site was generally less than 1 g/m^2; community biomass increased as much as 10 times between 1982 and 1986, with mean values ranging between 2 and 12 g/m^2. This was followed by an abrupt return to a concentration of <1 g/m^2 in 1987. Despite this change in biomass, the species composition of the zooplankton has generally remained the same from 1973 to 1988. The three dominant taxa are Daphnia, Diaptomus and Cyclops; rarer taxa include Bosmina, Epischura, Leptodora, Ceriodaphnia and Diaphanosoma. The abrupt increase and subsequent decline in zooplankton biomass may have been linked to the release of 250,000 kokanee fry in 1980.

Although all major zooplankton taxa increased in biomass from 1982 to 1986, Cyclops and Diaptomus have increased disproportionately; Daphnia, which composed almost 80% of the community biomass in the early 1970's, has accounted for less than 50%

of the total biomass since 1982, probably because of selective predation by kokanee. There has also been a notable decline in the Leptodora population, coinciding with a disproportionate increase in Bosmina. Epischura has become more common between 1982 and 1986, and both Ceriodaphnia and Diaphanosoma, have been found in higher densities in two more recent samples relative to samples taken before 1982.

Zooplankton appear to congregate between three and six metres depth during the open-water season, but are uniformly distributed in the water column from October to May. There is some suggestion that Daphnia undergo slight diel vertical migration within the surface 9 m during the summer. The horizontal distribution of zooplankton in Lake Koocanusa is also extremely heterogeneous, and this limits the accuracy of the present sampling program of monthly vertical hauls from a single station.

The fish community of Lake Koocanusa has undergone three major changes in species composition since the reservoir was impounded in 1972. Two new species, kokanee and yellow perch, have been unintentionally introduced; two other species, peamouth and northern squawfish, have increased in abundance; and four species, mountain whitefish, rainbow trout, westslope cutthroat trout and redside shiner, have decreased relative to their initially high abundances just after impoundment in the mid-1970's.

Lake Koocanusa does not appear to have experienced a "trophic upsurge" in the years following closure of the dam. Phytoplankton and zooplankton production was quite stable during the first decade, and the elevated nutrient levels in the early years can be attributed to upstream effluent loadings. Only fish populations showed the typical pattern of increase and decline as the reservoir aged, and this was probably a result of expanding lacustrine habitat, rather than augmented levels of nutrients and organic matter.

The Canadian portion of Lake Koocanusa is oligotrophic, verging on mesotrophic. Based upon chlorophyll and Secchi depth data, reservoir trophic state appears to have remained stable over the entire period since filling, nothwithstanding a sharp decline in phosphorus levels. There is a gradient in trophic state with distance down the reservoir: the Forebay area is clearly oligotrophic.

10.0 RECOMMENDATIONS

1. There is a need to resolve the discrepancy between U.S.G.S. and B.C. Environment measurements of nitrogen in Koocanusa Reservoir. This should be done through a careful, double-blind comparison of prepared samples of known nitrogen content by both agencies, and with split samples of Koocanusa water from the Border and Forebay sites. The objective would be to discover which set of data is the more accurate, determine if the error is additive or multiplicative, and devise a correction to allow use of a merged data set with confidence.

2. The extreme spatial heterogeneity of zooplankton populations in the reservoir calls into question the efficacy of the present sampling regime of a once monthly vertical haul at a single station. A more thorough program of sampling needs to be initiated, that would integrate simultaneous data from a number of points along the length of the reservoir. This would produce an estimate of the degree of horizontal variations (the largest unsampled component of variance), along with improved population density estimates. Such information may help to resolve the apparent anomalies in the present data.

3. The trends in Elk River water quality related to coal mining activities in the basin indicate that continued monitoring at an intensive level is warranted. The relatively stable water quality condition in the Kootenay River and Lake Koocanusa (with the exception of zooplankton) over recent years means there is no need to intensify monitoring of these systems.

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APPENDIX

CHEMICAL ANALYSIS PROTOCOL

Table A1 Detection Limits (mg/L) for Metals, Nutrients, Color and NFR Analyses on Lake Koocanusa, Elk River and Kootenay Rivers.

Constituent	Detection Limit	Time Period	Analytic Method
As	0.25	1982 – 1987	0040
	0.005	1972 – 1987	EQ01
	0.001	1985 – 1987	0180
	0.0001	1985 – 1989	X167
Cd	0.01	1982–1989	0040
		1987 – 1989	
	0.001	1985 – 1986	X167
	0.0005	1986 – 1988	X167
	0.0001	1986 – 1989	0060
Cu	0.01	1977 – 1984	EQ01
	0.01	1982 – 1989	0040
	0.001	1977 – 1984	EQ01
	0.001	1977 – 1989	0060 & X167
	0.0005	1985 – 1989	X167
Fe	0.1	1974 – 1982	EQ01, 0020, 0200
	0.01	1973 – 1989	0040
	0.01	1985 – 1986	X167*
	0.0005	1985 – 1989	X166*
Mn	0.02	1978 – 1982	0020
	0.02	1978 – 1979	0200
	0.02	1978 – 1979	EQ01
	0.01	1982 – 1989	0040, X167
	0.001	1983 – 1989	0060, X167
Pb	0.01	1982 – 1984	0040
	0.001	1973 – 1989	0060
	0.001	1985 – 1986	X167
	0.0007	1986 – 1989	x167
Zn	0.01	1978 – 1989	0040
	0.001	1973 – 1978	01 (4 different methods)
	0.005	1974 – 1989	0060
	0.005	1984 – 1986	X167
	0.0003	1986 – 1989	X167
TN	0.02	1986 – 1988	X183
	0.02	up to 1986	calculated
	0.001	1985 – 1988	X184

Table A1 Cont'd

Constituent	Detection Limit	Time Period	Analytic Method
NO2 + N03	0.02	up to 1989	1350
NH3	0.01	up to 1974	EQ01
	0.005	1974 - 1979	1351
TP	0.003	up to 1984	EQ01
	0.003	1984 - 1989	1390, X185
SRP	0.003	up to 1984	EQ01
	0.003	1985 - 1989	1380
Colour	5	1972 - 1977	EQ01
	5	1977 - 1989	1310
	5	1982 - 1989	1320

Note: The database documentation indicates method X167 for iron to have a D.L. of 0.0005 but the data for Elk R. suggest it to be 0.01. The reverse is true for X166.

Table A2 Analytic Methods

Database Analytic Code	Description
EQ01	Data that have been posted from the discontinued EQUIS data base, and for which exact analytic protocols are not clear
METALS	
0040	NHO3 digestion, Inductively Coupled Plasma arc spectroscopy (ICP) Used for quick scans of a sample
0060	HNO3 digestion, High temperature Graphite Analysis (Graphite furnace or HGA)
0180	HNO3 digestion, Hydride ICP
0200	20% A.R. digestion. Atomic Absorption (AA) analysis
X166	HNO3 digestion, AA analysis
X167	HNO3 digestion, Solvent Extraction, AA analysis
COLOR	
1320	Spectrophotometer, one wave length
1310	Spectrophotometer
EQ06	EQUIS Data – probably Hellige Aqua Tester
NUTRIENTS	
1380	Auto Ascorbic Acid
1390	Dig. Auto. Ascorbic Acid
X185	Dig. Auto. Ascorbic Acid
1351	Automated Bertholot Method
1350	Auto. Cadmium Reduction
1360	Block Dig. Auto. Color
X183	0.45 u; Auto. UV Dig.; Colormetric
X184	(GF/F); Auto. UV Dig., Colormetric